Executive summary

Introduction

This report describes the volume and characteristics of UK-affiliated research evidence on sustainable soil management for the delivery of three key ecosystem services (ES), namely: food production; water and nutrient cycling; and climate change mitigation. It focuses on soil quality and management on agricultural land and peatlands; and aims to identify evidence gaps and priorities for new research to guide future soils policy.

Sustainably managed soils are essential for the future delivery of a variety of ecosystem services (ES). However, the ability of soils to deliver these services is threatened by degradation processes such as erosion, compaction, loss of organic matter and acidification. UK governments are committed to promoting sustainable soil management to secure the delivery of essential ES for future generations. However, to achieve this aim there is a need to better understand the range of ES that soils provide and the extent to which soils can be effectively managed to ensure sustained provision of these services.

This review has been carried out as part of Soil Security Programme (SSP) co-ordination activities. The SSP aims to carry out research and knowledge exchange activities to help improve understanding of how soil systems respond to changes in management and climate. A key objective of the SSP is to gain an integrated and predictive understanding of the ability of soils to (i) perform multiple functions and (ii) resist, recover and adapt to perturbations. There were therefore clear synergies between Department for Environment, Food and Rural Affairs (Defra) and SSP objectives that could be met through a review of evidence from UK soils research and the development of a set of recommendations to inform future soils research.

Objectives

The main objective of the project was to determine the volume and characteristics of the evidence base available from UK-affiliated research organisations to address specific research questions on sustainable soil management and to provide an overview of the evidence. The specific objectives were to:

(i) Identify research papers and projects that address specific research questions
(ii) Provide an overview of what the evidence base indicates in relation to the research questions posed in the review
(iii) Identify evidence gaps and priorities for new research

The review focused on three key ES and five themes. The key ES were:

i. Food production
ii. Water and nutrient cycling
iii. Climate change mitigation
The five key themes covered were:

i. **Soil quality indicators (SQI’s) and ES**
   - the relationship between SQI’s and the delivery of ES

ii. **Soil degradation**
   - the impact of soil degradation processes on ES delivery

iii. **Sustainable soils**
   - the characterisation of sustainably managed soils

iv. **Soil management practices**
   - how soils respond to soil management

v. **Economic value of ecosystem services**
   - valuing soil functions and soil ES

**Methodology**

Research papers and projects that addressed the research questions were identified using a systematic search and review methodology. The evidence review process used aspects of a Quick Scoping Review and a Rapid Evidence Assessment, including the development of mainly open-ended, non-impact research questions. Keywords were used in a series of ‘Web of Science’ searches restricted to papers from UK-affiliated research organisations published since 1995 and quick searches of relevant government and research organisation websites. Three project workshops delivered in Edinburgh, Aberystwyth and Reading in July 2015 provided an opportunity for researchers, advisers and other relevant stakeholders to provide feedback on draft review outputs and to validate the research gaps identified.

**Results**

The initial searches resulted in 1,563 papers related to the 23 research questions. In a first screening, titles were checked for relevance to specific research questions and in a second screening abstracts were assessed for relevance resulting in the identification of 559 research papers in total. Each of these papers referenced multiple other papers that contributed to the research area. The quick searches identified 474 Defra projects and 95 BBSRC/NERC projects of relevance to sustainable soil management and the ES covered here.

**Conclusions and recommendations**

**Search methodology**

The systematic search and review methodology had certain advantages over other approaches in terms of transparency and lack of bias and was effective in finding a broad range of key research papers that addressed the research questions. However, there were a number of important papers and projects that were only identified through the workshops. The material found through the workshops was to some degree dependent on the individuals that were able to attend and their particular areas of expertise/interest. Interviewing a short list of experts identified during the systematic search stage, while introducing a degree of bias, may have resulted in the rapid identification of key research papers and projects across the five themes.
SQI’s and ES

The evidence review indicates that the relationship between SQI values and ecosystem service delivery is still poorly understood. Thresholds and workable ranges for physical and chemical indicators have been developed, but not necessarily related to the delivery of specific ES. Soil biological indicators require further investigation to better understand their capacity to reflect the degree of soil degradation or the ability of soils to perform specific functions. The development of reference values for particular soil type, land use and climate combinations could help provide a useful framework against which soil quality could be assessed. Soil monitoring should focus on determining the rate of change in soil properties and change in the proportion of soils within particular SQI value ranges, as such changes could have implications for soil functioning and the delivery of key ES. Further detailed investigations at long term experiments will be crucial for determining the relationship between SQI values, soil management and ES delivery; and randomised and replicated field experiments will also be important to link soil properties to soil function.

Other priorities for further research include:

- The development of robust SQI measurement protocols and the standardisation of indicators; to include intensive measurements on arable, grassland, upland and lowland sites to cover the main agricultural land uses, agro-climatic zones and soil types in the UK.
- Investigation into the nature of C and nutrient cycling in a range of soil types in productive agricultural systems and the long term effects of agricultural inputs on these soil processes; including the implications of manufactured fertiliser use for C sequestration and storage and the balance between climate change mitigation considerations and food production.
- The functional ecology of arbuscular mycorrhizal fungi (AMF) and other soil biota in different agricultural systems and possible synergies and antagonisms with a need for increased food production.
- The importance of fungal diversity in C and nutrient cycling.
- Better understanding of functional diversity and functional redundancy within different agricultural systems; and of links between soil biodiversity, functional diversity and soil morphology at various scales.
- Investigation of observational (e.g. earth observation, mapping, sensors) and analysis technologies (e.g. modelling, big data and analytics) to determine their potential to improve the implementation and monitoring of sustainable soil management in the UK.
- The potential use of public participation in the measurement of certain suitable SQI’s.
- Better understanding of the key factors and processes influencing C fluxes under different soils (particularly peatlands) and land management practices.

Soil degradation

For soil degradation, few studies quantified linkages and thresholds between the change in soil properties and associated changes in soil processes. Unless changes in soil properties can be linked to soil processes and functions, it is difficult to understand and predict the impact of soil degradation in a meaningful way. Nevertheless, it is clear that policies that maintain or enhance SOM contents should result in multiple benefits for society.
In terms of future work, field monitoring of soil erosion should be continued and the efficacy of mitigation options assessed in high risk areas. The impact of soil compaction on ES is also poorly understood and further work is needed to assess compaction avoidance and mitigation techniques for integration into farming systems. Soil organic matter decline has significant implications for soil function and climate change mitigation and there is a need to understand carbon (C) cycling dynamics better, particularly in organo-mineral and peat soils. The impacts of changing soil acidity and N-deposition on ES in the uplands (and other sensitive habitats such as lowland heath) also needs further research.

Other priorities for further research include:

- Better understanding of the degree to which soil erosion needs to be controlled to maintain soil functions and soil natural capital and the types of soil management practices that need to be encouraged in particular circumstances.
- Improving the modelling of soil erosion rates and sediment yield to take account of the typical magnitude and variability of observed soil erosion rates.
- The development of numerical models of soil formation to help implement soil erosion mitigation strategies at appropriate spatial scales.
- The identification of erosion, compaction, organic matter loss and acidification ‘tipping points’ for soil function.
- More field research to determine the typical range of effectiveness values for mitigation strategies to help ecosystems recover from soil degradation; and to understand how rates of soil erosion, %SOC etc. impact on soil functions and the delivery of ES.
- More detailed monitoring of soil C stocks for different land use, management practices and land use changes to provide further insight into the potential changes in sequestered C.
- More research on a wider range of environmental conditions to further develop and differentiate the ‘SOC management range’ concept.
- Monitoring of soil compaction (soil structure and bulk density) and soil erosion (soil depth and soil erosion features) to determine whether any future changes in the nature and timing of rainfall, and its relationship with soil management practices, is affecting the severity of these degradation processes.
- Assessing the impact of N deposition on species-rich grassland (and other habitats) under different management strategy scenarios.

Sustainable soils

The evidence highlights the importance of protecting SOC and conserving soil biodiversity for multiple benefits, including sustainable (profitable) food production, C and nutrient cycling and climate change mitigation. Policies that encourage protecting, maintaining and enhancing soil C levels will be essential in the future to optimise the provision of food, energy and water, regulation of climate and maintenance of biodiversity.

For most ES, we do not have a clear idea of what properties a sustainable soil should have in terms of (for example) SOC content, soil structure, nutrient content and metal concentrations, although past research outputs provide some guidance. Sustainability also
needs to be considered in terms of what ES should be prioritised in different parts of the landscape and what sustainability means for each ES in each location.

Other priorities for further research include:

- The better definition of different criteria for soil sustainability depending on the ES in question and to define concepts such as ‘soil health’ and ‘sustainable’ in order to set goals and objectives that can be used in policy and management development.
- More scientific evidence to define thresholds or critical levels for SOC and soil biodiversity for a wider range of soil type, agro-climatic and land use scenarios to help guide policy and provide more information on soil status.
- Further work on soil physics and soil engineering disciplines, as well as soil chemistry and biology, to support the development of soil compaction avoidance and mitigation options.
- The continued development of rapid methods of soil structural assessment, from practical visual evaluation methods that can be used in the field by practitioners to the development of new technology to provide research tools that are more effective at quantifying soil structure on a continuous scale. There is also a need for a standard method for assessing soil productivity across a range of scales.
- Further investigation of the implications of soil management practices, particularly the input of organic materials and manufactured fertilisers, for soil quality in general and soil metal concentrations in particular.
- The viability of setting targets for soil biology and biodiversity within a soil type, land use, climate and ES framework; including establishing acceptable rates of change in key soil properties such as bulk density, SOC and functional groups of soil biota.
- The merits and risks of earthworm introduction into fields and other forms of soil biota/food web manipulation.
- Developing a methodology to quantify the risk of harm to soil by soil organic matter decline and soil erosion.
- Investigating the possible expansion of soil sustainability models (such as MOSES - Modelling Soil Ecosystem Services - and the Critical Zone Observatory models) to include additional processes and functions to improve the simulation of multiple soil ES provision and the effects of external drivers such as climate change.
- The continued need for stakeholder engagement and interdisciplinary research to help address land-based challenges; including the need to report scientific research findings in a range of formats that are suitable for all stakeholders so that key groups can engage in discussion about soil management practices and sustainability.

Soil management practices

There is a wealth of information on how soil management practices impact on soil properties and crop production, but for other ES there is less information. Within this context, the importance of engagement between researchers and farmers was repeatedly emphasised at the project workshops as being a priority to ensure understanding of agricultural issues (e.g. practical and economic challenges) and to aid the effective implementation of new knowledge and technologies. We have a poor understanding of whether or not soil biodiversity can be manipulated to improve soil function. It is clear that soils have the capacity to store more C, but there are constraints to the ability of this approach to provide genuine C sequestration and climate change mitigation. However,
strategies that promote the maintenance or enhancement of SOC will generally improve soil properties for multiple benefits.

It is important that land management policies take account of the C sequestration potential of different soil types and land management practices in UK agro-climatic conditions while also recognising that policies encouraging the protection or enhancement of SOC represent a ‘win-win’ strategy in terms of climate change mitigation and the sustainability of agricultural systems. This also underlines the importance of supporting effective national soil monitoring networks for the assessment of soil C concentrations and stocks.

Adaptation strategies that increase the resistance and resilience of agricultural soils, such as maintaining field drainage systems on slowly permeable soils, growing cover crops, applying bulky organic manures and avoiding soil compaction also generally help improve farm profitability and should be encouraged through policy.

Priorities for further research include:

- The development of land management incentives such as payment for ecosystem services (PES) from government or through land tenure agreements that result in sustainable soil management practices and the maintenance of key soil functions on rented land.
- The development of practical tools for advisors and farmers to produce clear soil management advice for all agricultural production systems.
- To better understand the response of microbial communities to soil management/disturbance and to assess the relationship between microbial diversity and function.
- Assess the C sequestration potential of manipulating plant species diversity and associated soil biodiversity (e.g. microbial functional groups) in agricultural soils and implications for overall food production levels.
- Improve our understanding of how nitrous oxide emissions respond to soil management.
- Develop evidence-based soil management practices to meet the challenges of ‘sustainable intensification’; including new techniques to maintain food production with decreasing water resources.
- Initiate and maintain long-term field experiments, in a range of locations to investigate critical soil P and K concentrations required to optimise crop yields.
- Better understand the impacts of reduced and no-tillage practices on the delivery of key ES in the UK.
- Engage with the public concerning mitigation strategies as certain biological approaches may have negative effects on some ecosystem services and land use.
- Investigation into the social, economic and environmental viability of new climate change mitigation and adaptation techniques such as paludiculture.

**Economic value of ES**

The development of tools that incorporate the economic value of ES provided by soils is in its early stages. Only a few studies have attempted to value soil ES. However, there is potential for the valuation of soil to be used to develop decision support tools to support resource management and soil protection policy. The development of such valuation tools could help support policy mechanisms such as Payments for Ecosystem Services (PES). A
thorough assessment of the ‘natural capital’ provided by soils also needs further development; and to ensure that soils have sufficient protection within policy frameworks it will be important that the value of soils is considered separately from the value of land.

More work is needed on the costs of soil management interventions that go beyond the simple assessment of direct costs and include intangible and indirect effects. There is also a need for more information on societal benefits including the development of ecosystem damage costs for a wider range of pollutants, goods and services. New methods are required for assessing the full range of societal benefits derived from soil management interventions, from replacement cost to willingness to pay, stated preferences and the full market and non-market value of ecosystem goods and services.

It will be important to consider how ES concepts and societal benefit from soil management policies can be better communicated to the public so that policy is correctly understood and communication between different stakeholder groups is improved.

Other priorities for further research include:

- The development and testing of soil compaction avoidance and mitigation methods.
- Better understanding of the implications of soil compaction and mitigation options for soil processes and functions; and for flooding.
- Better understanding of the likely extent of changes to agricultural landscapes resulting from the increased production of energy crops and their impacts on ES such as nutrient cycling, C sequestration and flood control.
- How to value the ES provided by SOC and soil biodiversity.
- Better understanding of the recovery rates of soil C and the interactions between SOM, biodiversity, transformations of nutrients and soil structure.
- How to ensure that soil ES continue to be delivered into the future, i.e. that soils are managed sustainably.
- How to promote a diversity of agricultural systems by allocating land to the delivery of different goods and services according to its suitability.
- To better understand how soil management links to soil properties and soil function and the minimum standard of soil quality in terms of soil physical, chemical and biological properties and biological diversity (related to functional diversity and redundancy) needed to deliver the key soil ES.
- Better understanding of the impacts of structural changes in the livestock grazing sector on soil quality and function; including how grassland and grazing management affects soil characteristics and nutrient cycling.
- More interdisciplinary research to identify management regimes that support sustainable livestock production.
### Contents

- **Executive summary**
- 1. Introduction – Background and objectives
- 2. Methodology and scope
  - 2.2 Screening the results
  - 2.3 Quick searches
  - 2.4 Synthesis of evidence
  - 2.5 Project workshops
- 3. Soil quality indicators and ecosystem services
  - 3.1 Introduction
  - 3.2 Can we benchmark indicators for the delivery of ecosystem services or specific soil functions?
  - 3.3 How can indicator values be interpreted with respect to ecosystem service delivery?
  - 3.4 Are there any indicators that can detect change in ecosystem delivery within a policy cycle?
  - 3.5 What is the practicality of different approaches?
  - 3.6 Are there any established envelopes of normality for SQI’s?
  - 3.7 What do changes in soil biodiversity and soil C content mean for ecosystem service delivery?
- 4. Soil degradation
  - 4.1 Introduction
  - 4.2 At what point do degradation processes significantly affect soil quality and function?
  - 4.3 How is soil degradation best measured and can any tipping points for soil function and the delivery of the key ecosystem services be identified?
  - 4.4 How is climate change likely to affect soil degradation processes?
- 5. Sustainable soils / Aspirational soil quality targets
  - 5.1 Introduction
  - 5.2 What properties should a sustainable soil have?
  - 5.3 What targets should we have for soil quality?
  - 5.4 Is there a model for sustainable soils to deliver the key ecosystem services?
  - 5.5 What effect may climate change have on aspirational soil quality targets?
- 6. Soil management practices
  - 6.1 Introduction
  - 6.2 How should soil be managed to achieve sustainability?
  - 6.3 What do we mean by a sustainably managed soil and how do we measure it?
1. Introduction – Background and objectives
This report describes the volume and characteristics of UK-affiliated research evidence on sustainable soil management for the delivery of three key ecosystem services and aims to identify evidence gaps and priorities for new research to guide future soils policy. The report focuses on agriculturally managed soils and peatlands and also summarises the state of knowledge in five key areas of soils research to support policy.

Soil is the central component of terrestrial ecosystems and plays a primary role in all global biogeochemical cycles. Sustainably managed soils are essential for the future delivery of a variety of ecosystem services, such as food production, water management, climate change mitigation and supporting biodiversity. However, the ability of soils to deliver these services is threatened by degradation processes such as erosion, compaction, loss of organic matter and acidification, and it is imperative that soils are managed to counter these threats and protect the soil resource so that the natural capital (the capacity of an ecosystem to provide beneficial goods and services) of soil ecosystems can be preserved for future generations (McBratney et al., 2014).

The commitment of UK governments to promoting sustainable soil management is clear from statements made in “The State of Scotland’s Soil” report (Dobbie et al., 2011), the “Soil Monitoring Action Plan for Scotland” (Black et al., 2013), the “Welsh Soils Action Plan” (Welsh Government, 2008) and “Towards a Land Strategy for Northern Ireland” (Miller et al., 2015). In England, through the Natural Environment White Paper, Defra is committed to ensure that all soils are managed sustainably and degradation threats tackled successfully by 2030, in order to improve the quality of soils and to safeguard their ability to provide essential ecosystem services and functions for future generations. Similarly, a key aim of the Environment Agency’s “Collaborative Research Priorities 2015-19” is to better understand how soil and water resources are affected by changing agricultural practices and how government can best work with farmers and other land managers to achieve sustainable agricultural production. The Environment Agency is also actively investigating how soil and land management practices can be used to deliver flood risk management benefits. However, to achieve these aims there is a need to better understand the range and importance of the ecosystem services that soils provide and the extent to which soils can be effectively managed to ensure sustained provision of these services. There is also a significant challenge to define suitable metrics of sustainable soil management (if this is possible) and to prioritise appropriate soil management interventions.

This review has been carried out as part of Soil Security Programme (SSP) co-ordination activities. The SSP aims to carry out research and knowledge exchange activities to help improve understanding of how soil systems respond to changes in management and climate. This SSP has responded to the need to gain an integrated and predictive understanding of the ability of soils to (i) perform multiple functions and (ii) resist, recover and adapt to perturbations. There were therefore clear synergies between Defra and SSP objectives that could be met through a review of evidence from UK soils research and the development of a set of recommendations to inform future soils research. In particular, there was a need for the review to address commitments made on soil management made in the Natural Environment White Paper.
This report presents the results of a review of evidence to support policy on sustainable soil management. The review focused on three key ecosystem services and five themes. The key ecosystem services were:

(i) Food production – the ability of soils to support the sustainable production of crops and livestock and associated nutritious food in sufficient quantity and quality to support the human population

(ii) Water and nutrient cycling – the ability of soils to regulate, store and provide water and nutrients, mitigate flooding and cycle nutrients *in situ* (NB this does not include the mitigation of diffuse water pollution from agriculture)

(iii) Climate change mitigation – the ability of soils to store carbon (C) and prevent enhanced emissions of nitrous oxide (a greenhouse gas with a global warming potential c. 300 times greater than carbon dioxide)

The five key themes covered (i) the relationship between soil quality indicators (SQI’s) and the delivery of ecosystem services, (ii) the impact of soil degradation processes on ecosystem service (ES) delivery, (iii) the characterisation of sustainably managed soils, (iv) how soils respond to soil management (in terms of their ability to deliver ES’s) and (v) the economic values of the ecosystem services provided by soil.

Soil management was considered for all three broad soil types, namely mineral, organo-mineral and peaty:

- **Mineral** (\(<100 \text{ g kg}^{-1}\) soil organic matter – SOM for soils with >50% clay in the mineral fraction; \(<60-100 \text{ g kg}^{-1}\) SOM on a sliding scale of % clay in the mineral fraction for soils with <50% clay) and **organic mineral** soils (60-250 g kg\(^{-1}\) SOM) are defined as soils with less than 200-250 g kg\(^{-1}\) SOM content (depending on the % clay in the mineral fraction) in the surface horizon (otherwise referred to as topsoil)
- **Organo-mineral** soils are defined as soils that have a topsoil relatively rich in organic matter but with a peaty surface layer less than 40 cm thick (Defra SP1106)
- **Peaty** soils are defined as soils with a topsoil organic matter content greater than 200-250 g kg\(^{-1}\) (depending on the % clay in the mineral fraction)

The main objective of the project was to determine the volume and characteristics of the evidence base available from UK-affiliated research organisations to address specific research questions on sustainable soil management in agricultural soils and rough grazing peatlands (forestry and urban soils were excluded) and to provide an overview of the evidence. The specific objectives were to:

(iv) Identify research papers and projects that address specific research questions

(v) Provide an overview of what the evidence base indicates in relation to the research questions posed in the review

(vi) Identify evidence gaps and priorities for new research

A systematic process, described in section 2, was used to identify research papers and projects that respond to the research questions under each theme and to capture the key information. Sections 3 to 7 present the findings of the search for evidence, including an
overview of the volume and characteristics of the overall evidence base, and a synthesis of the evidence for each of five key themes and 24 related research questions. Section 8 sets out the evidence gaps and priorities for future research within the context of encouraging sustainable food production (and delivering the objectives of the Food 2030 Strategy), protecting ecosystem services and supporting a strong rural economy resilient to climate change. Finally, section 9 provides a summary and conclusions.

The research questions were related to five themes:

1. **Soil quality indicators (SQIs) (including physical, chemical and biological indicators) and ecosystem services**
   - Can we benchmark indicators for the delivery of ecosystem services or specific soil functions?
   - How can indicator values be interpreted with respect to ecosystem service delivery?
   - Are there any indicators that can detect change in ecosystem delivery within a policy cycle?
   - What is the practicality of different approaches?
   - Are there any established envelopes of normality for SQI’s?
   - What do changes in soil biodiversity and soil C content mean for ecosystem service delivery?

2. **Soil degradation**
   - At what point do degradation processes significantly affect soil quality and function?
   - How is soil degradation best measured and can any tipping points for soil function and the delivery of the key ecosystem services be identified?
   - How is climate change likely to affect soil degradation processes?

3. **Sustainable soils**
   - What targets should we have for soil quality?
   - What properties should a sustainable soil have?
   - Is there a model for sustainable soils to deliver the key ecosystem services?
   - What effect may climate change have on aspirational soil quality targets?

4. **Soil management practices**
   - How should soil be managed to achieve sustainability?
   - What do we mean by a sustainably managed soil and how do we measure it?
   - What level of intervention is needed to reach the target of sustainably managed soil?
   - How does soil management impact on soil quality?
   - How do different soil types respond to management practices and do they still have capacity to store more soil C?
   - Can soil biodiversity be manipulated to improve ecosystem service delivery?
   - How may climate change affect the choice of soil management measures to achieve sustainability?

5. **Economic value of ecosystem services**
   - What is the economic value of the key ecosystem services provided by sustainably managed soils?
- Do we fully understand how soil management relates to soil function to accurately quantify this?
- What work has been done to assess the relative cost and societal benefit of soil management interventions?

In each section the volume and characteristics of the overall evidence base are presented; the objectives and findings from principal projects and research papers outlined; and the implications for policy and further research summarised.
2. **Methodology and scope**

To identify research papers and projects that addressed the research questions a systematic search and review methodology was used to minimise bias and provide transparency. The evidence review process used aspects of a Quick Scoping Review and a Rapid Evidence Assessment, including the development of mainly open-ended, non-impact research questions with use of keywords in a series of searches.

Primary themes and secondary questions were agreed and refined with the Project Steering Group (including representatives from Defra and the Environment Agency) at the project inception meeting. A protocol was developed to define the key ecosystem services to consider, the key themes for research questions, keywords, scope (geographic range, time period and language) and strategy for extracting information using study matrices by theme. The development of the protocol and keywords was an iterative process taking account of comments and discussions at Steering Group meetings.

The ‘Web of Science’ was used to carry out searches for research papers published since 2000. ‘Web of Science’ is a web-based scholarly research database and search facility that provides access to bibliographic information such as the Science Citation Index (http://apps.webofknowledge.com). Combinations of keywords, search strings and boolean operators were used to reduce bias and provide more focused and productive results. The geographical range was limited to the United Kingdom and European affiliated research papers. However, where research gaps were identified this was expanded to international papers to estimate the volume of evidence available internationally.

The database and search terms used, along with the number of hits and a full list of the evidence produced was recorded (Appendices I-VII). Two screenings were then carried out to refine the search results and identify research papers relevant to the review themes and questions:

**2.2 Screening the results**

The first screening used the title of the evidence to check its relevance to either answering the research question or identifying the work that is needed to answer the research question. In some cases, the evidence questioned the relevance of the research question.

The second filter used the full abstract to check for relevance and to select research papers that would be used in the synthesis stages of the review. This resulted in a final list of research papers of relevance to each research question (Appendices I to V).

**2.3 Quick searches**

To identify relevant projects that addressed the research questions and themes a number of quick searches were carried out using keywords and search facilities available from:

- Defra
- Biotechnology and Biological Sciences Research Council (BBSRC)
- Natural Environment Research Council (NERC)
- Economic and Social Research Council (ESRC)
- Forestry Research
- Scottish Government
The list of Defra, BBSRC and NERC projects that resulted from these searches is presented in Appendix VIII.

2.4 Synthesis of evidence
Information from the papers of greatest relevance was extracted into study matrices that summarised:

- The specific research question to which the research paper related
- The ecosystem services considered
- The soil types covered
- The information of relevance to addressing each research question (e.g. evidence of impact-response measured or observed)

The study matrices, reference lists and quick search results were then used to provide an overview of the volume and characteristics of the overall evidence base, and synthesise the evidence in terms of what the evidence indicates in relation to each research question posed. The implications of the findings for future policy and/or practice were then considered with suggestions for further research.

2.5 Project workshops
Three project workshops were delivered in Edinburgh, Aberystwyth and Reading in July 2015 to provide an opportunity for researchers, advisers and other relevant stakeholders to provide feedback on draft review outputs and validate the research gaps identified. This was carried out through presentation of the draft review results and structured break-out sessions.
3. Soil quality indicators and ecosystem services

3.1 Introduction
This section covers soil physical, chemical and biological indicators and the extent to which they can be used to assess the ability of soils to deliver key ecosystem services. Many studies have considered soil quality indicators for general soil quality and the statistical assessment of the variation in SQI values to produce ‘concern’ and ‘action’ levels (e.g. the UK Soil Indicator Consortium projects), but few studies consider the relationship between SQI values and the delivery of specific ecosystem services and the identification of any functional thresholds.

Twenty one research papers addressed the question of benchmarking indicators and interpreting them with respect to ecosystem delivery (Figure 1); 6 papers covered the use of indicators to detect change in ecosystem delivery within a policy cycle; 12 covered the practicality of different approaches; and 13 provided information on envelopes of normality for SQI’s. Finally, 16 papers investigated how changes in soil biodiversity and soil C content can affect ecosystem service delivery.

![Figure 1. Number of papers identified as relevant to addressing specific research questions within the “soil quality indicator and ecosystem services” theme.](image-url)
3.2 Can we benchmark indicators for the delivery of ecosystem services or specific soil functions?

Volume and characteristics of the overall evidence base

The search and screening process resulted in 36 research papers that specifically address the benchmarking of indicators for the delivery of ecosystem goods and services. However, these 36 papers reference many more related documents and sources. Of the 36 papers identified, 13 explicitly covered food production, 20 water and nutrient cycling and 7 climate change mitigation (Table 1). However, in many cases the ecosystem services that the papers addressed was not specified in the abstract or keywords.

A number of researchers (e.g. Archer et al., 2013; Ball et al. 2007; de Bello et al., 2010; Storkey et al., 2013) investigated the relationship between ‘short lists’ of specific SQI’s or ‘functional traits’ and the delivery of ES, ranging from a single indicator of flooding risk (Archer et al., 2013) to 18 physical and chemical indicators for crop productivity (Armenise et al., 2013). By contrast, Feld et al. (2009) examined 531 indicators of ES delivery reported in 617 peer-reviewed journal articles between 1997 and 2007.

Other papers (e.g. Banwart et al., 2012; Menon et al., 2014; Smit et al., 2012; Usher et al., 2006) described initiatives to develop better understanding of how soils deliver ES and the specific properties, processes and functions that are critical to the sustained delivery of goods and services. Included in this was the concept of the soil life cycle from soil formation to productive use and soil degradation and the need to develop a better understanding of how soil properties and processes change at various points in this ‘cycle’ (Banwart et al., 2012).

Finally, some soil scientists (e.g. Nortcliff, 2002; Kibblewhite, 2008) have questioned whether it is appropriate to pursue the notion of a limited number of soil properties to indicate ES delivery, arguing that ‘soil health’ is an integrative property that reflects a variety of biological processes provided by a diversity of soil organisms under the influence of soil physical and chemical properties.

Table 1. The number of papers that investigated the relationship between soil quality indicators (SQI’s) and ecosystem service delivery by SQI type (physical, chemical and biological).

<table>
<thead>
<tr>
<th>Indicator type</th>
<th>Food production</th>
<th>Water and nutrient cycling</th>
<th>Climate change mitigation</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>Physical</td>
<td>10</td>
<td>13</td>
<td>5</td>
<td>20</td>
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<td>Chemical</td>
<td>8</td>
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<td>Total</td>
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<td>20</td>
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<td>36</td>
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</tbody>
</table>

Most papers were general in nature and did not target specific soil types (Table 2). Even for single site studies it was not always possible to determine the general soil type from the information available in the paper abstract. Most lowland studies covered mineral soils and studies focusing on upland grassland soils (e.g. Usher et al, 2006) were more likely to include
organo-mineral/peaty soils. Defra project SP1106 (“assessment of the response of organo-mineral soils to change in management practices”) is a key information source for organo-mineral soils and covered:

- The occurrence of organo-mineral soils in England and Wales.
- Soil degradation pressures associated with organo-mineral soils.
- Expert elicitation to assess the ecosystem services delivered by each of ten land-cover types.
- The impact of projected future climate change on organo-mineral soils.
- Fifteen best practices for maintaining or increasing soil C storage in organo-mineral soils.
- The potential impact of changing to ‘best practice for retaining C in organo-mineral soils’ on various ecosystem services.
- Knowledge gaps and uncertainties related to the management of organo-mineral soils.

Table 2. The number of papers that investigate the use of soil quality indicators for ecosystem service delivery in mineral, organo-mineral and peaty soils specifically.

<table>
<thead>
<tr>
<th>Mineral</th>
<th>Organo-mineral</th>
<th>Peaty</th>
</tr>
</thead>
<tbody>
<tr>
<td>7</td>
<td>1</td>
<td>2</td>
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</table>

*What does the evidence base indicate in relation to the question posed?*

Since 2000, a considerable amount of work has been carried out to develop a minimum dataset (MDS) of indicators to help policy decisions on soil quality and management. This began with an Environment Agency-led project on the “Identification and development of a set of national indicators for soil quality”. Loveland *et al.* (2002) and Nortcliff (2002) explained how it was difficult to identify quantifiable criteria for soil quality, partly due to the wide range of uses to which soils are put and also because of the complexity of the soil system and the possibility that changes in the soil may be slow and possibly only occur when some threshold is reached. Nortcliff (2002) argued that any index of soil quality must consider soil functions, which are varied and often complex. A soil which is considered to be of high quality for one function may not be so for other functions. As a consequence there are potentially many soil properties which might serve as indicators of soil quality, and research is required to identify the most suitable. Nortcliff (2002) also stressed the importance of standard methods when selecting SQI’s and, comparing SQI values.

Science Project SC030265 on “The development and use of soil quality indicators for assessing the role of soil in environmental interactions” (Merrington *et al.*, 2006) built on the assertions of Loveland *et al.* (2002) that soil quality indicators (SQI’s) should be linked to soil function and later to ecosystem services. A ‘challenge process’ based on work by Sparling *et al.* (2003) assessed eleven potential indicators (nine recommended by Loveland *et al.*, 2002 and two others not formerly identified) in terms of their relevance, interpretation, measurability and cost. ‘Trigger values’ or ‘workable ranges’ for each indicator were initially established using statistical assessment of the range of values within baseline datasets rather than the establishment of thresholds beyond which any particular soil function is impaired. The ‘trigger values’ were intended to be used to build up a ‘weight
of evidence’ to determine if there was a potential issue with soil quality and its ability to perform specific ‘functions’ such as food and fibre production or environmental interaction.

In the final report, trigger values for some indicators such as soil pH and soil bulk density (BD) were developed, “whereas for others such as SOC, alternative approaches were suggested such as the use of reference sites or ‘no reduction values’ under specific site conditions”. The MDS of SQI’s for ‘environmental interaction’ included SOC, total nitrogen (N), Olsen P, available and total copper (Cu), nickel (Ni) and zinc (Zn), BD and soil pH. The project (SCO30265) also established approaches to determine ‘critical changes’ in these indicators to trigger further ‘action’ (i.e. management and/or policy intervention).

Soil physical indicators
The MDS included SOC (which can be viewed as chemical, biological and/or physical), six predominantly chemical variables and a single soil physical property (BD). The use of physical soil properties as indicators of soil quality was therefore further investigated in Defra project SP1611 (“Indicators of soil quality – physical properties”), which used a ‘logical sieve’ approach (Black et al., 2003; Ritz et al., 2009) to assess the relative suitability of each potential physical SQI for national scale soil monitoring. The ‘logical sieve’ approach involves the scoring of indicators by experts and stakeholders against a range of scientific and technical criteria such as whether or not the indicator is useful in detecting meaningful change, objective, transparent and reproducible. Each soil physical property was scored in terms of:

- Soil function – does the candidate SQI reflect all soil function(s)?)
- Land use - does the candidate SQI apply to all land uses found nationally?
- Soil degradation - can the candidate SQI express soil degradation processes?
- Does the candidate SQI meet the challenge criteria used by Merrington et al. (2006)?

SP1611 also used power analysis in an attempt to understand the variability of indicators and to determine the ability to detect a particular change in an SQI at a particular confidence level. However, the project concluded that “the evidence base for analysing the candidate SQIs is poor: data are limited in spatial and temporal extent for England and Wales, in terms of a) the degree (magnitude) of change in the SQI which significantly affects soil processes and functions (i.e. ‘meaningful change’), and b) the change in the SQI that is detectable (i.e. what sample size is needed to detect the meaningful signal from the variability or noise in the signal)”. This supported conclusions by Merrington et al. (2006) that for some SQI’s it is extremely difficult to establish trigger values or benchmarks for different soil types and land uses and that the use of reference sites and ‘no increase/decrease values’ under specific site conditions may be more useful.

Use of the ‘logical sieve’ approach and statistical analysis resulted in the selection of 7 physical SQI’s: depth of soil; soil water retention characteristics; packing density (related to BD and clay content); visual soil evaluation; rate of erosion; sealing; and aggregate stability. All the physical SQI’s presented strengths and weaknesses (e.g. soil depth and surface sealing are regarded by many as indicators of soil quantity rather than quality), and the evidence base for each varied in its degree of certainty, variability and extent such that priority could not be given to any particular indicator or group of indicators.
Other researchers that have investigated the use of physical SQI's to assess ES delivery include Archer et al. (2013) who related field saturated hydraulic conductivity (K-fs) values to areas of surface runoff generation and associated flood risk on a hillslope in the Scottish Borders. Acreman & Holden (2013) demonstrated differences in flood functions both within and between wetland types and investigated how physical soil characteristics relate to the delivery of water cycling (flood reduction) services. Ball et al. (2007) investigated the relationship between soil physical properties and both crop production and climate change mitigation.

Banwart et al. (2012) measured soil physical and chemical properties at four key sites that represent various points along the soil life cycle. Information on soil degradation and changes in biogeochemical properties were used to develop a model of soil development that combined process descriptions of carbon (C) and nutrient flows, a simplified description of the soil food web, and changes in soil structure and soil aggregation. The researchers argue that the quantification of soil processes could help improve understanding of what properties and processes are needed for soil ecosystem services delivery, including the quantitative monetary valuation of these services within the soil life cycle.

Palmer and Smith (2013) developed a framework for linking soil structural quality with surface runoff. They carried out a survey of soil structural condition in South-West England, and linked this to the generation of surface runoff. Soil structural condition was assessed following guidelines outlined by Hodgson (1997) and a classification of soil structural condition in relation to surface runoff was developed based on work by Holman et al. (2001, 2003). Overall, it was found that 38% of the 3,243 surveyed sites had soil structural degradation sufficient to produce features of enhanced surface runoff. Sites classed as having High or severe soil degradation mostly occurred on land used to grow potatoes, maize or winter cereals. Land used to grow late harvested crops (e.g. potatoes and maize) displayed the most damage, with the occurrence of highly or severely degraded soil being c.15% higher compared to winter cereals (mean occurrence of highly or severely degraded soil was more than 60% on winter cereals).

Soil biological indicators
Since the 1990s, soil biological indicators have been assessed in an increasing number of field trials and monitoring programmes across Europe (Pullman et al., 2012) and were further investigated in the UK under Defra projects SP0529 (SQID: Soil quality indicators – developing biological indicators) and SP0534 (Scoping biological indicators of soil quality - phase II). Ritz et al. (2009) used the ‘logical sieve’ approach to select biological indicators of soil quality, for national-scale soil monitoring. They argued that soil biota play a fundamental role in the majority of ecosystem services, and so soil biological properties are logical candidates as effective indicators, to complement soil physico-chemical properties. Experts and stakeholders scored 183 candidate biological indicators using the ‘logical sieve’ approach, which resulted in a ranked list of 21 indicators covering a range of genotypic-, phenotypic- and functional-based indicators. However, they acknowledged the inherent sensitivity of the indicators to multiple environmental factors and the need for further methodological development, standard operating procedures, ability to discriminate between soil/land use combinations, and ecological interpretation.
Defra project SP0534 investigated the ability of a number of biological indicators to detect change in soil quality due to three significant pressures in UK soils, namely: nitrogen deposition, heavy metals from sludge applications to land and habitat restoration after mining. The project concluded that there was no universal indicator (measure) or method that provided sensitivity to a range of contrasting pressures, and that “a suite of soil biological methods would be a more informative approach to monitoring changes in soil biological status where multiple pressures are at play, or where the pressures influencing soil are unknown”. Sensitivity analysis indicated that the suite should include, as a minimum: Phospholipid fatty acid analysis (PLFAs), Terminal restriction fragment length polymorphism (TRFLP - for fungi, bacteria and archaea), Multiple substrate induced respiration (MSIR) and microarthropods. Similarly, to monitor differences and interpret status and changes in soil biological quality due to changes in land use, an appropriate suite of biological indicators would include: PLFAs, TRFLP (for fungi and archaea), MSIR and multi-enzymes.

A number of other research papers have investigated the relationships between soil biota, soil quality and ES delivery. A’Bear et al. (2014) reported on interactions between spatially-separated aboveground and belowground biota and their importance for food production (principally pollination and biological control) and C and nutrient cycling. They describe how feedbacks between aboveground and belowground biota influence ecosystem services such as nutrient cycling and plant growth (food production). For example, aboveground herbivory generally stimulates nutrient cycling by decomposers; and root herbivory and mycorrhizal association both appear to increase floral attractiveness to insect pollinators. However, it is not clear how important these interactions and feedbacks are or how effectively they operate within different productive agricultural systems.

Using new data combined with previously published data, Webster et al. (2001) found that there was a relationship between the size of the soil microbial community (microbial biomass) and the quality of soil C as a resource for microorganisms.

Legay et al. (2014) focused on factors that influence nutrient cycling in grassland soils. They investigated the relative contributions of soil abiotic properties and above-ground and below-ground plant traits, to variations in microbial processes involved in grassland N turnover. Ecological traits are the “morphological, physiological, behavioural or life-history attributes of organisms” (Pulleman et al., 2012). Below-ground plant traits were the most relevant traits for explaining variation in community structure and abundances of soil microbes involved in nitrification and denitrification. They argue that consideration of below-ground plant traits, increases our ability to describe variation in the abundance and the functional characteristics of microbial communities in grassland soils.

Cole et al. (2006) explored abundance, species richness and functional diversity of soil biota in an upland grassland soil (Sourhope series; a freely draining, mineral, brown forest soil) particularly with respect to C and N cycling. They concluded that functional redundancy (duplication of functional roles among soil biota) at the species level is most likely to occur in species rich faunal groups with generalist feeding behaviour. The paper also investigated the land management factors that influence earthworm, enchytraeid, collembolan and mite diversity, their competitive interaction and their influence on soil properties. However,
Prendergast-Miller et al. (2008) found that enchytraeid worms were not useful indicators of ammonia deposition or soil acidification in an ombotrophic bog.

De Bello et al. (2010) propose the use of functional traits in a range of organisms including plants and soil invertebrates as indicators of ecosystem service delivery, specifically food production and nutrient cycling. They reviewed 247 studies and argued that clusters of traits of plants and soil organisms are essential for the delivery of nutrient cycling, herbivory, and fodder and fibre production services. They argue that the assessment of trait-service clusters will represent a crucial step in ecosystem service monitoring, including nutrient cycling and fodder/fibre production and in balancing the delivery of multiple, and sometimes conflicting, services in ecosystem management.

Feld et al. (2009) reported that, despite great effort to develop indicator systems over the past decade, there is still a considerable gap in the widespread use of indicators for many of the multiple components of biodiversity and ecosystem services. They examined 531 indicators reported in 617 peer-reviewed journal articles between 1997 and 2007. They stress the need to develop common monitoring schemes within and across habitats. “Filling these gaps is a prerequisite for linking biodiversity dynamics with ecosystem service delivery”. However, the paper does not specify which ecosystem services are best assessed using biological indicators, but states that “the functional, structural and genetic components of biodiversity are poorly addressed despite their potential value across habitats and scales”.

Feld et al. (2010) discuss the development and application of suitable indicators for ecosystem assessment and delivery. They tested 24 indicators of ecosystem service delivery using seven criteria. The seven criteria included the purpose of the indicator, direct/indirect linkages to ecosystem services, spatial scale and scalability across scales, applicability of benchmarks/reference values, and availability of data and protocols.

Storkey et al. (2013) investigated plant and soil invertebrate traits as biological indicators of the ability to support crop production through soil biodiversity and pollination. They described how quantification of the functional links between arable plants and their associated invertebrate consumer communities will be the first step in extending the trait-based approach to quantifying trade-offs and synergies in ecosystem service delivery from grassland systems to arable landscapes.

Usher et al. (2006) described the origins, development and characteristics of a major programme of research into soil biodiversity, the NERC Thematic Research Programme ‘Biological Diversity and Function in Soils’. The programme was conceived to address a number of questions relating to the role of biodiversity in the ecological functioning of soils. It aimed to characterise the roles played by all major groups of soil biota in the C cycle; and to determine the extent to which reductions in soil biodiversity may limit its ability to perform essential ecosystem services.

The research was focussed on a single, upland grassland site (Sourhope Research Station) and combined both field studies and experiments in controlled laboratory conditions. The main achievements of the programme included understanding the outstanding diversity of
soil biota. Usher et al., (2008) argued that the research made substantial advances towards our understanding of both the extent and function of the biological diversity of soil ecosystems and acknowledged that similar work needs to be carried out in other land management situations to determine whether or not soil biological processes differ between contrasting soil management regimes.

Related to 'Biological Diversity and Function in Soils', the on-going Wessex BESS (Biodiversity and ES for Sustainability) is a 6-year (2011-2017) NERC funded research project, which aims to understand how biodiversity underpins ecosystem services and functions at a range of spatial scales.

**Multiple indicators and soil monitoring programmes**

Other studies have investigated multiple physical and chemical variables (SQI’s) to develop a soil quality index. For example, Armenise et al. (2013) used 18 chemical and physical soil properties to assess soil quality for crop productivity and yet concluded (due to a lack of a statistically significant correlation between the soil quality index and yield) that other soil quality indicators, not measured in the study, were more influential upon yield in the experimental year.

Defra project SP1611 noted that integrating more than one SQI (including physical, chemical and biological indicators) to produce a ‘so-called’ soil quality score for particular soil functions is likely to only propagate errors. By contrast, other researchers have shown how compound indicators of soil quality can be derived from physical, chemical and biological soil parameters using multivariate analyses and can enable monitoring of temporal change in the delivery of different soil ecosystem services (Ruiz et al. 2011; Velasquez et al., 2007). Nevertheless, Defra project SP1611 concluded that the relationship between SQIs, soil functions and the delivery of ecosystem goods and services is complex. Indeed, “important gaps remain in the realisation of a conceptual model for these inter-relationships, let alone their quantification. There is also a question of whether individual quantitative SQIs can be related to ecosystem services, given the number of variables”.

Smit et al. (2012) described a number of recommendations for the development of a General Surveillance system for soil ecosystem service delivery. The system would make use of existing networks such as the soil sampling infrastructure provided by The Dutch Soil Quality Network (DSQN). The paper also discussed the development of a stakeholder participation model involving three groups (land users, soil scientists and policy makers) to determine what degree of change in ecosystem delivery is acceptable.

Menon et al. (2014) provide an overview of the European Commission-funded research project on Soil Transformations in European Catchments (SoilTrEC). The SoilTrEC project (http://www.soiltrec.eu/main/aimsObjectives.html) aims to quantify the processes that deliver soil ecosystem services and to quantify the impacts of environmental change on key soil functions. The project also includes the economic valuation of soil ecosystem services and the impact of soil degradation on the delivery of key soil functions (section 7).
Soil quality indicators and assessments of ‘soil health’
Kibblewhite et al. (2008) proposed that soil quality and sustainability is dependent on the maintenance of four major functions: C transformations; nutrient cycles; soil structure maintenance; and the regulation of pests and disease (Figure 2). They also stressed the importance of quantifying the flow of energy and C between these four major functions as an essential task for the assessment and management of soil quality and sustainability. They present soil quality or ‘soil health’ as an integrative property that reflects the capacity of soil to respond to agricultural intervention, so that it continues to support a wide range of ecosystem services; and conclude that “measurement of individual groups of organisms, processes or soil properties does not suffice to indicate the state of the soil health”. Nortcliff (2002) also questioned whether it is appropriate to pursue the notion of a limited number of soil quality indicators.

Karlan et al. (2003) summarised how the soil quality concept has developed overtime and that the focus is upon two broad areas 1) to educate and 2) assessment. Soil quality indicators have therefore been promoted for use by farmers and advisers as useful tools to assess soil quality. For example, the United States Department of Agriculture (USDA) have produced a number of factsheets for physical, chemical and biological soil quality indicators, describing how they can be used to assess ‘soil health’. Each factsheet discusses factors which affect the specific indicator, how the indicator relates to soil function and methods for improving soil quality: http://www.nrcs.usda.gov/wps/portal/nrcs/main/soils/health/.

The USDA have produced a literature review (USDA, 2015) consisting of c.180 peer-reviewed articles investigating the impact of conservation practices upon soil chemical and physical properties relevant to soil health. The USDA also aim is to collect information related to biological properties and the economics of conservation practices.

Discussion
There are hundreds of SQIs that could potentially be used to assess the delivery of soil ecosystem goods and services, although recent studies have reduced ‘useful’ physical, chemical and biological indicators to around 10-20 in each case. For some indicators there is sufficient data and model development to link values to the delivery of specific ecosystem services across a number of land uses, soil types and management conditions (e.g. relating saturated hydraulic conductivity to surface runoff generation and flooding risk). However, for many others, particularly biological indicators, this work still needs to be done.

SQI values that indicate ability to deliver one ES may not indicate the ability to deliver another public good or ES. For example, measures of soil biodiversity that are sufficient to indicate good soil structure for water regulation may not be sufficient to preserve the full range of soil biota that could provide future products to the pharmaceutical industry. However, for the ecosystem services considered in this review soil properties and processes that indicate the sustained delivery of one ES (e.g. food production) on the whole correlate well with delivery of the others (water and nutrient cycling and climate change mitigation). Pursuing agricultural production in the short term can result in soil degradation that limits the ability of soil to perform other functions. However, the sustainable management of agricultural soils, including maintaining or enhancing of SOC levels can provide multiple benefits (Banwart et al., 2015).
While the measurement of SQI values is important to establish a baseline (i.e. the current state of soil), it is the rate of change in soil properties and change in the proportion of soils within particular SQI value ranges that is more important; and the implications of these changes in terms of soil processes and functioning that are key to effective soil monitoring.

Clearly, benchmarking indicators for the delivery of ES or specific soil functions is challenging, but previous projects have demonstrated that it is possible to develop trigger values and threshold values for different soil types and land uses that indicate that management or policy intervention may be needed. For example, Merrington et al. (2006) developed ‘concern’ and ‘action’ values for soil bulk density that varied according to SOM content.

3.2.1 Implications for policy and further research

Soil monitoring should focus on determining the rate of change in soil properties and change in the proportion of soils within particular SQI value ranges, as such changes could have implications for soil functioning and the delivery of key ES.
It is important that information linking soil properties to soil function is developed in randomised and replicated field experiments. Long term experiments are also crucial for determining the relationship between SQI values, soil management and ES delivery. Detailed investigation such as that carried out at the Sourhope Research Station (as part of the NERC project on “Biological Diversity and Function in Soils”), Defra’s Soil QC (SP0530) and Long-term Biosolids sites (SP0143) should be extended to other sites to improve our understanding of C transformations; nutrient cycles; soil structure maintenance; and the regulation of pests and disease (i.e. soil health and sustainability) and how these relate to ES delivery.

To improve our understanding of soil health and sustainability and how to measure them the following research areas should be explored:

- Investigation into the nature of C and nutrient cycling in a range of soil types in productive agricultural systems and the long term effects of agricultural inputs on these soil processes. Better understanding of soil C dynamics will be key to achieving sustained delivery of soil ecosystem services.
- More field research to relate soil management to SQI values and ES outcomes (e.g. crop quality and yield in the case of food production) and to develop effective conceptual models of the relationship between SQIs, soil functions and the delivery of ecosystem goods and services across a range of soil types, land uses and management scenarios and at different points in the soil life cycle (formation, productive use and degradation).
- An important step in the process will be to validate the conceptual models using new and existing experimental evidence.

3.3 How can indicator values be interpreted with respect to ecosystem service delivery?

*Volume and characteristics of the overall evidence base*

The search and screening process resulted in 22 research papers that discussed the interpretation of indicator values for ecosystem service delivery. However, these 22 papers reference many more related documents and sources. Of the 22 papers identified, 13 covered food production, 14 water and nutrient cycling and 6 climate change mitigation (Table 3). Fifteen papers focused on biological indicators, three on chemical indicators and two on physical indicators.

Table 3. The number of papers that address how indicator values can be interpreted with respect to ecosystem service delivery by SQI type (physical, chemical and biological).

<table>
<thead>
<tr>
<th>Indicator type</th>
<th>Ecosystem service</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Food production</td>
<td>Water and nutrient cycling</td>
</tr>
<tr>
<td>Physical</td>
<td>2</td>
<td>-</td>
</tr>
<tr>
<td>Chemical</td>
<td>2</td>
<td>2</td>
</tr>
<tr>
<td>Biological</td>
<td>9</td>
<td>14</td>
</tr>
<tr>
<td></td>
<td>13</td>
<td>17</td>
</tr>
</tbody>
</table>
The majority of papers (18) covered SQI’s in mineral soils with only two explicitly covering peaty (or organo-mineral) soils.

Table 4. The number of papers that investigate how indicator values can be interpreted with respect to ecosystem service delivery in mineral, organo-mineral and peaty soils specifically.

<table>
<thead>
<tr>
<th>Mineral</th>
<th>Organo-mineral</th>
<th>Peaty</th>
</tr>
</thead>
<tbody>
<tr>
<td>19</td>
<td>-</td>
<td>2</td>
</tr>
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</table>

Thirteen papers covered the relationship between biological indicators and nutrient cycling, particularly in grassland soils (e.g. Bardgett & McAlister, 1999; De Vries et al., 2011; Grigulis et al., 2013).

Only two papers specifically investigated the relationship between soil physical indicators and crop production as an ecosystem service (Mueller et al., 2009; 2010).

What does the evidence base indicate in relation to the question posed?

Physical indicators

Defra projects SP1611 (Indicators of soil quality – physical properties) and SP1305 (Cost curve for mitigation of soil compaction) provide useful information on the relationship between soil physical properties, soil management and the delivery of a wide range of ES including crop production, water regulation and climate change mitigation. However, the searches carried out in this project identified few papers that specifically focus on interpreting physical indicator values for ES delivery.

Mueller et al. (2009) tested the diagnostic value of different visual soil evaluation methods to assess soil structural quality in relation to the measurement of soil physical properties and yield of cereals. Measured soil physical properties and crop yields correlated significantly with visual soil evaluation scores (Figure 3). Unfavourable visual structure was associated with increased dry bulk density, higher soil strength and lower infiltration rate but correlations were site-specific. Results showed that shape and size of aggregates were quickly recognizable diagnostic features of soil structure. Biological features like earthworm or root numbers were less reliable indicators of soil structure than aggregate characteristics. Mueller et al. (2009) concluded that visual soil structure assessment is a useful diagnostic tool and can provide a clear indication of soil condition and potential crop yield.

In 2010, Mueller et al. (2010) analysed available methods for assessing the productivity function of soils with the aim of developing a global reference framework. They found that soil structure was a crucial criterion of agricultural soil quality and methods of visual soil evaluation (e.g. Peerlkamp, 1967; Diez and Weigelt, 1997; Shepherd, 2000, 2009; Werner and Thaenert, 1989; Ball et al., 2007) were powerful tools for recognising dynamic agricultural soil quality and controlling soil management processes at field scale. They concluded that visual evaluation approaches have potential to be integrated into an assessment framework of the soil's productivity function, and that such a framework could serve as a reference base for ranking soil productivity potentials on a global scale.
Figure 3. Peerlkamp structure scores (or “notes”) versus relative cereal grain yield at sites D (in Germany) and E (in Canada). Absolute grain yields ranged from 7.7 to 8.8 t ha\(^{-1}\) at site E and from 7.5 to 9.2 t ha\(^{-1}\) at site D.

A number of research projects are working towards better interpretation of SQI’s for ES delivery. For example, the GESSOL projects (*fonctions environnementales du sol – GESTion du patrimoine SOL*, i.e. soil environmental functions – soil heritage management [www.gessol.fr](http://www.gessol.fr)) have contributed to the development of soil quality indicators (Pulleman et al., 2012). One project on the effects of soil compaction resulted in the identification of threshold void volumes in the soil (e.g. 0.15 m\(^3\) m\(^{-3}\) in loamy soils) beyond which the performance of crops and hydrodynamic properties can be affected.

**Chemical indicators**

The relationship between soil chemical properties, such as soil pH and extractable nutrients, and soil functions such as crop production and nutrient cycling is well established (e.g. Defra, 2010; Johnston, 2001; Johnston *et al.*, 2001a, 2001b; Johnston and Dawson, 2005). The crop production function is also closely linked to climate change mitigation through net primary productivity (NPP) and partitioning between the harvestable crop, roots and root exudates for C sequestration and storage. However, further advances are being made into the more efficient use of non-renewable resources such as phosphorus and the development of new chemical indicators to assess crop production and nutrient cycling potential.
For example, Hart and Cornish (2014) investigated the use of an adjusted soil P test to determine the risk of P pollution from grassland soils. The soil P test was determined from the measured Colwell-P minus the threshold P (an indicator of P availability to a growing crop) estimated from a P-buffering index (PBI). They found that Colwell-P and soil PBI together provided a widely applicable test to assess the risk of total and dissolved reactive P loss from grassland soils with good vegetation cover. They concluded that the same threshold values could be used for both food production and nutrient cycling/buffering ecosystem services.

Rowe et al. (2014) found that an indicator of phosphorus (P) availability (bicarbonate-extractable P) was a more widespread constraint to the productivity of semi-natural ecosystems in the UK than was N availability. Furthermore, the stock of mineralisable N was much less well able to explain variation in a metric of productivity (Ellenberg N score - E-N), particularly in extensively-managed habitats.

**Biological indicators**

Up until 2000, interpretation of biological soil indicators in terms of ES delivery was largely based on expert judgements. A more robust and quantitative approach was needed requiring empirical testing and the development of models. Soil biodiversity datasets could provide potentially important sources of information to explore general relationships between soil characteristics, the structure (or trait) of the soil community, ecosystem functioning and ES delivery (Mulder et al., 2011). For example, the UK Countryside Survey has measured soil invertebrate diversity (using Tullgren funnel extractions to 8 cm soil depth) in 1998 (1,286 main plot samples in 256 km squares) and 2007 (927 main plot samples in 238 km squares), indicating an overall significant increase in total invertebrate catch in samples from all habitats (Emmett et al., 2010).

Ecological traits such as soil invertebrate number and diversity could be used to link soil degradation and other human-induced perturbations to effects on soil functions and ES delivery. For example, polymerase chain reaction (PCR)-based nematode detection is a robust and affordable quantitative tool, but information on how values relate to soil processes, functions and ES delivery remain to be developed (Van Megen et al., 2009). This is an extremely challenging task as ecosystem services operate at different spatial and temporal scales (Dominati et al., 2010). Nevertheless, the commercial application of soil quality tests/indicators is an emerging area. For example, NRM Laboratories now offer the ‘Soil Health’ analytical package, which provides an index of soil health by measuring soil nutrient status, soil texture, organic matter and biological activity (measured by the Solvita® test or CO₂ burst) (NRM Soil Health Booklet).

A better understanding of soil biodiversity at specific sites and in long-term trials or observatories could help identify tipping points for soil processes, functions and ES delivery. Such work could help determine the appropriate balance between functional diversity and functional redundancy in different circumstances (Are some agro-ecological systems sustainable with a relatively low level of functional redundancy? i.e. soil functions performed adequately by relatively few species) and the level of soil biodiversity required to impart resistance and resilience within soil systems (Breure et al., 2012).
Pulleman et al. (2012) provided an overview of current knowledge on the characterization and assessment of soil biodiversity. They provided examples of biological soil indicators and monitoring approaches and discussed the value of databases for developing a better understanding of the relationship between soil management, soil functions and ecosystem services. They concluded that integration of monitoring approaches and data sets offers good opportunities to better understand the relationship between biological indicators and ES delivery and the application of such knowledge by land managers and policy makers. They emphasise the importance of developing indicator reference values that take account of different combinations of land use, soil type and climate (Pulleman et al., 2012). Such references do not yet exist for most indicators at a European scale, although for biological indicators density ranges for different groups of organisms have been published for a selection of soil and land use types in the Netherlands (Rutgers et al., 2009) and France (Cluzeau et al., 2012).

Each individual soil cannot provide all functions to an optimal level simultaneously, and so preference needs to be given to certain ecosystem goods and services depending on the context (Haygarth & Ritz 2009). This will potentially require offsetting some functions at the expense of others in some localities, and the spatial management of soil systems at local, regional and national scale.

Bardgett and Cook (1998) reported that low-input agricultural grassland systems promoted greater soil biodiversity and favoured fungal-pathways (rather than bacteria-dominated decomposition pathways) with a more heterogeneous habitat and resource leading to domination by more persistent fungal-feeding fauna. However, further research was needed to test the hypothesis that low input grassland farming and enhanced soil biodiversity is positively associated with soil ecosystem stability and self-regulation of ecosystem function.

Bardgett and McAlister (1999) reported that across a wide range of meadow grassland sites in northern England, along a gradient of long-term management intensity, fungal:bacterial biomass ratios (measured by phospholipid fatty acid analysis; PLFA) were consistently and significantly higher in the unfertilised than the fertilised grasslands. However, it was found that microbial activity, measured as basal respiration, did not differ between the sites and, following the re-instatement of low-input management on an improved mesotrophic grassland, that neither the cessation of manufactured fertiliser applications nor changes in cutting and grazing management significantly affected soil microbial biomass or the fungal:bacterial biomass ratio. Nevertheless, Bardgett and McAlister proposed that a significant increase in the soil fungal:bacterial biomass ratio, and perhaps total microbial biomass, may be useful indicators of successful conversion to a grassland system reliant on ecosystem self-regulation rather than on manufactured fertilisers and pesticides.

De Vries et al. (2011) reported from two greenhouse experiments using intact soil cores that higher fungal biomass can be considered as an indicator of higher N retention in soils due to higher rates of immobilisation and lower N losses through denitrification.

A few papers have investigated the use of decomposition rates as indicators of soil quality and ES delivery. For example, Griffiths et al. (2001) studied the functional stability of the short-term decomposition of added plant residues. Substrate mineralisation kinetics were a
good indicator of hydrocarbon pollution, and functional stability, particularly resistance (the ability of the soil to withstand the immediate effects of perturbation; in this case heat and copper stresses). They were able to quantify differences between paired soils with: high or low plant species diversity; hydrocarbon pollution or not; and high or low output agricultural management practices. However, further work was needed to develop the technique in relation to ES delivery and soil health.

Hauser et al. (2005) used the rate of decomposition of Senna spectabilis leaves to distinguish between different land uses (undisturbed fallow of about 4 years, young secondary regrowth of about 12 years and secondary forest of at least 25 years) and proposed that the method could be developed as one component of a soil quality or soil function indicator if decomposition rate could be linked to other soil properties, crop yields on agricultural land and biomass accumulation rate in forest.

Godley et al. (2004) considered soil microbial biomass quotient (biomass expressed as a percentage of the total SOC) and respiration quotient (the specific rate of carbon-dioxide production by soil microbes) as indicators of stress on soil microbial populations, for example by toxicity from high soil metal concentrations. They reported that variations in biomass quotient and respiration quotient could be sufficiently large to make interpretation of their values, as a response to stress, problematical, since the indicators are affected by a wide variety of other factors such as the biodegradability of soil organic C amendments, plant inputs of organic C, and natural variations in microbial population sizes and the rhizosphere with depth.

Biomonitors are organisms that provide quantitative information on environmental quality and the ability of ecosystems to deliver goods and services. A few papers investigated the influence of plants and their interaction with soil microbial communities on ES delivery. For example, Madejon et al. (2006) defended the use of plants as biomonitors, and argued that they have important advantages over soil analyses as indicators of soil quality, particularly when investigations are made on a large scale.

In a study to determine relationships between ecosystem properties, plant traits and soil community characteristics at three grassland sites in different parts of Europe, Grigulis et al. (2013) found that exploitative plant species and soil microbial communities dominated by bacteria, with rapid microbial activities, were linked with greater fodder production, but poor C and N retention. Conversely, dominance by conservative species (with lower specific leaf area, leaf N concentrations and higher leaf dry matter content) and soil microbial communities dominated by fungi, and bacteria with slow activities, were usually linked with low production, but greater soil C storage and N retention. They concluded that managing grasslands for selected, or multiple, ecosystem services will require a consideration of the joint effects of plant and soil communities and that further understanding of the mechanisms that link plant and microbial functional traits is essential to achieve this.

Bommarco et al. (2013) acknowledged that research efforts and investments are particularly needed to reduce existing yield gaps between low input agricultural systems and systems reliant on manufactured fertiliser and pesticides by integrating “context-appropriate
bundles of above and below ground ecosystem service-providing organisms” into crop production systems.

Fagan et al. (2010) focused on ecosystem function and habitat restoration rather than the delivery of ecosystem services, but in an assessment of calcareous grassland restoration found that the ant species *M. sabuleti* was a good indicator species for calcareous grassland restoration success and, alongside information from the plant community, could increase the confidence with which restoration success is judged.

Some researchers have investigated the use of soil enzymes as early indicators of a change in soil function. For example, Turner et al. (2002) found that substrate induced activity of the soil enzyme p-glucosidase was positively correlated with clay, total C and microbial C contents. This indicated that substrate-induced p-glucosidase activity is an integrative measure of physico-chemical and biological soil properties and may have applications in monitoring biological soil quality. However, the findings were not explicitly related to ES delivery.

Defra project SP1601 investigated approaches to influence soil biota communities to enhance delivery of improved soil fertility and quality. It was concluded that systems-oriented approaches that involve the provision of energy-containing substrate (principally organic matter) or optimisation of the soil habitat have greater potential than ‘point interventions’ that target specific, often monotonic, aspects or sub-components of soil biotic assemblages. Controlling tillage practices was suggested as one way of optimising the soil habitat by avoiding excessive disturbance of the soil, and encouraging biological mechanisms to improve soil structure. The project concluded that “definition of the precise ‘architectural configuration’ of any particular soil system - or indeed soil systems in general - that would reflect an optimal state is reasonable but not yet feasible, and is a key research requirement”.

Finally, Buckland et al. (2012) reviewed methods for quantifying the biodiversity of regions, and considered issues that should be addressed in designing and evaluating a regional monitoring scheme. They produced a practical guide to the types of survey that would be appropriate for addressing different objectives for biodiversity monitoring.

**Peatlands**

Defra project NE0141 (Towards the development of a UK Peatland Code) investigated the development of indicators to monitor and quantify the climate regulation benefits and other (e.g. water quality, biodiversity, aesthetic) co-benefits of peatland restoration. They consider different approaches to quantify GHG emission reductions following peatland restoration, including ‘tier 1’ methods based on default values for different land uses; ‘ tier 2’ methods that include country specific emission factors and other data; and ‘ tier 3’ methods that involve more complex, model-based approaches that require empirical measurements to calibrate coefficients to new sites. One ‘ tier 2’ type approach that provided cost-effective proxy variables for assessing C fluxes is the Greenhouse gas Estimation Site Types (GEST) approach, which classifies vegetation assemblages according to their relationship with water table levels, while also considering nutrient status, soil pH and land use; and provides a
more detailed assessment of GHG emission and C storage than current default values (Couwenberg, 2011).

Worrall *et al.* (2008) considered long term records of dissolved organic carbon (DOC) flux from two catchments with peat-covered headwater and compared both annual and monthly DOC flux records with a range of hydroclimatic indicators to test which component of droughts may contribute to increasing DOC flux. The most important variable for explaining the DOC flux was the runoff from the catchments overlying a seasonal cycle and an underlying upward trend was present in some records. The lack of any evidence for any additional biogeochemical reactions associated with drought supported evidence that DOC loss from peat is limited by its solubility and that its production is fast on the time-scale of runoff events.

Artz *et al.* (2008) reported that the activity and functional diversity of the microbial community in five European peatlands responded to vegetation succession. The community-level physiological profile (CLPP – a measure of the respiratory response of the soil microbial community to ecologically relevant substrates) provided vital information about the relative importance of different plant functional types on potential rates of labile C turnover and C storage, particularly in the early stages of regeneration of cutover peatlands. However, comparison of data over a number of years indicated that species composition tends to be relatively stable, but microbial abundance and biomass were more variable depending for example on weather conditions and crop rotations (Bispo *et al*., 2009; Pérès *et al*., 2011; Rutgers *et al*., 2009).

### 3.3.1 Implications for policy and further research

Despite significant progress with interpreting indicator values, major scientific and practical issues remain to be addressed. These include the development of indicator reference values for different combinations of land use, soil type and climate; and obtaining a better predictive understanding of the relationships between soil degradation, soil biodiversity and ES delivery (Pulleman *et al*., 2012).

Interpretation of indicator values in terms of ES delivery is a first step in using indicators in soil monitoring programmes to support decision making by land managers and policy makers. It will therefore be important to:

- Develop indicator reference values that take account of different combinations of land use, soil type and climate (Pulleman *et al*., 2012).
- Identify how indicator values change over time, since a detectable change in indicator values may indicate soil degradation and a trend towards loss of function.
- Provide sufficient robust soils data for different land use, soil type and climate combinations to determine a meaningful degree of difference in indicator values that represents a measureable change in terms of ES delivery.

Fitting indicator values to soil function performance and transforming the specific units of the indicator to a uniform scale for ES delivery is a remaining challenge for many indicators (EEA, 2011), particularly emerging biological indicators.
It is important to address the balance between food production and the provision of other ES goods and services. There is strong evidence that to truly optimise the soil physical conditions and soil biota communities for particular ES will require context-dependent approaches. Preference will need to be given to certain ES, depending on the context, since each soil cannot provide all functions to an optimal level simultaneously.

At the workshops the importance of developing practical tools for advisors and farmers was emphasised. There is a need to build on successful projects such as the UK Agriculture and Horticulture Development Board (AHDB) “Healthy Grassland Soils” guide (http://dairy.ahdb.org.uk/technical-information/grassland-management/healthy-grassland-soils/#.VehcXxGYbIX) and produce clear soil management advice for all agricultural production systems.

Improving understanding of how soils function and how indicators relate to ES delivery under different management regimes will require further field-based experimentation, long-term sites and model development. This will include better understanding of:

- How soil physical properties including visual evaluation and the use of image processing to assess soil structure at meso-and micro-scales relates to soil function, the diversity of soil biota and the delivery of ES
- Soil biodiversity and its importance for soil processes and functions for different land use, soil management and soil type combinations
- Functional diversity and functional redundancy within different agricultural systems
- Links between soil biodiversity, functional diversity and soil morphology at various scales

3.4 Are there any indicators that can detect change in ecosystem delivery within a policy cycle?

Volume and characteristics of the overall evidence base

The search and screening process resulted in only six research papers that specifically addressed the question of detecting change in soil ES delivery within a policy cycle or legal frameworks for soil protection. Of the six papers identified, four covered food production, six water and nutrient cycling and two climate change mitigation (Table 5). Three papers addressed physical indicators, four chemical indicators and six biological indicators.

Table 5. The number of papers that investigate whether or not soil quality indicators (physical, chemical and biological) can detect change in ecosystem delivery within a policy cycle.
What does the evidence base indicate in relation to the question posed?

Changes in soil properties occur slowly and are not easily measurable within a few years. Nevertheless, Defra project SP1605, sub-project F explored the use of “outcome-focused” indicators of soil quality that could be compatible with policy reporting cycles. They concluded that “modelling outcomes from data describing relative rates of soil forming and degrading processes may be useful in certain circumstances, although this is dependent on the availability of data and the precision of models. There is more scope for establishing indicators for the outcomes of locally-targeted actions, such as within catchments, than nationally”.

From the search methodology used in this report, a number of papers reported on the assessment of SQI’s against certain criteria, but few considered the time scale of change in relation to policy cycles. For example, Aalders et al. (2009) evaluated SQI’s in terms of their applicability against a number of important environmental and logistical parameters, including relevance to different soil types, functions, habitats and threats to soil, the inherent variability of soil, and a range of technical aspects such as analytical complexity, precision and reproducibility of analytical results and whether a standard operating procedure (SOP) existed for the technique. However, the ability of indicators to detect change within a policy cycle was not considered as it was recognised that changes in soil quality usually occur over a minimum time frame of 10-15 years.

Bone et al. (2010) described a definition of soil quality incorporating soil’s ability to meet multifunctional requirements and provide ecosystem services. They identified the need for a method to assess soil quality that is not function dependent, but uses a restricted set of indicators that are cross-functional (i.e. applicable to many soil functions). They proposed a ranking based approach to soil quality assessment to systematically prioritise geographical areas at risk from soil degradation where detailed investigation is required, and argued that the method could be relatively quick, easy and cost effective. They also suggested the use of pedotransfer functions (i.e. empirical equations that estimate soil property values, such as conductivity and strength, based on fundamental properties such as bulk density, texture and SOM content) as a reference for comparison with actual measurements. For example, the use of soil type, topsoil textural class, land use, and mean temperature to estimate topsoil OC could provide a predictive guideline of what the actual SOC value should be in any given location. Finally, they proposed that a better understanding of the relationship between soil organisms, soil properties and soil quality could help identify organisms/species/SQI’s that can be used as holistic indicators of soil quality or of more specific threats such as soil contamination. To develop an effective soil quality monitoring framework there is an ongoing need to establish linkages between soil indicators.

Hirsch et al. (2009) reported on changes in SOC and the abundance and diversity of microbial communities under three contrasting management regimes: grass, arable crops and bare fallow with regular ploughing. After fifty years, SOC, light fraction (low density) organic matter, total microbial biomass, the abundance of bacterial communities and the abundance and diversity of mesofauna (mites and collembola) were all much lower in the bare fallow. However, the bacterial diversity was similar in the bare soil to that in the grass or arable crops receiving fresh plant inputs. This demonstrated the stability of the soil bacterial community in terms of its diversity and indicated that soil microbial community
size may be a better indicator of soil quality (alongside assessments of soil structure as the two have been related – Six et al., 2006) than bacterial diversity.

By contrast, Bending et al. (2000) reported that the metabolic profiles of soil microbial communities could have potential for use as early indicators of the impact of management on soil functioning and soil quality. After 16 months of contrasting cropping (a continuous grass-clover ley and a rotation of vetch, spring barley and clover) there were no changes to soil biomass N, microbial respiration, total OM, light-fraction OM, labile organic N or water-soluble carbohydrates. However, the metabolic profiles of soil microbial communities were found to be highly sensitive to management practice. They concluded that patterns of microbial substrate utilization and metabolic diversity were more sensitive to the effects of management than are OM and biomass pools, and therefore may have value as early indicators of the impacts of management on soil biological properties, and hence soil quality. The indicators changed within sixteen months of the treatment, which has positive implications for the ability to detect change within a policy cycle. However, it was not possible to relate the changes in microbial metabolic profiles and metabolic diversity to specific soil functions or ecosystem services.

Garcia-Ruiz et al. (2008) discussed the use of soil enzymes as early indicators of soil quality change under contrasting agricultural management practices. In 18 paired olive orchards in southern Spain under contrasting organic and conventional management, loss on ignition was higher (greater organic amendment) and available inorganic N lower (manufactured N fertiliser was not applied) in organically managed soils, and soil enzyme activities were higher when considered across all sites ($P<0.01$). They highlighted the need for extensive comparative assessment to draw clear conclusions concerning changes to soil quality under contrasting soil management practices. Tillage intensity was an overriding factor in affecting soil properties and loss on ignition was able to detect change in soil quality as well as measurement of enzyme activity. However, the changes were not related to soil function or ES delivery, and although information on time since organic accreditation was captured for each site the paper did not relate this to the ability to detect change within policy cycle time scales.

Zagal et al. (2009) argued that labile soil C fractions and enzymatic activity are typically more sensitive to changes in soil management practices or environmental conditions than total SOM, and consequently are well-placed as early indicators of change in soil quality. Their results on volcanic soils confirmed the value of labile soil C fractions and dehydrogenase activity as sensitive indicators for detecting changes in SOM in the short term both for annual arable crop and crop-pasture rotations. Microbial biomass and dehydrogenase activity were the indicators most sensitive to changes in soil-crop management. However, the important link to soil function and performance was not made. The use of labile soil fractions as sensitive indicators of changes in SOM was also explored in Defra projects SP0310 (To develop a robust indicator of soil organic matter status) and SP08014 (Soils within the Catchment Sensitive Farming Programme: Project to deliver improvements in soil management).

Kibblewhite et al. (2012) stated that better soil monitoring information is needed to support targeting and evaluation of soil protection measures and that more reliable information is
required about the effectiveness of soil protection measures for different soils under different land use and management scenarios. However, time scales for detecting change in soil quality were not a prime consideration.

Although a number of soil quality indicators have direct relevance to soil and environmental policy, since they can be related to soil processes, soil functions and ES delivery, meaningful and detectable change in most SQIs is out of step with any single 5-year soil policy cycle, making it difficult to link particular changes in SQIs to particular policy activities (Defra project SP1611). Few indicators are able to respond to management changes within a policy cycle of five years. This presents challenges in detecting trends that can feed into short-term policy development or to gauge the effectiveness of soil protection policies.

Soil properties are influenced by many factors and it is not usually possible to link particular changes in SQI values to particular policy activities, not least because physical changes in soil quality usually occur more slowly than any developments in soil protection policy. However, soil monitoring can be used to determine the overall effectiveness of soils policies in reducing soil degradation, even if it cannot be used to evaluate the effectiveness of individual measures.

3.4.1 Implications for policy and further research

Soil monitoring (the regular measurement of robust soil indicators to determine the degree of temporal change in soil properties and functions) is essential to determine the overall effectiveness of soil protection policy even if it cannot detect meaningful change within a policy cycle.

Field-based experiments that improve our understanding of how soil properties and function respond to soil management and long-term experiments that can help characterise the relationship between soil biota communities, soil structure and ES delivery are crucial to provide robust scientific evidence for the development of key SQI’s for better soil quality monitoring in the future. A better understanding of the relationship between soil organisms, soil properties and soil quality will help identify organisms/species/SQI’s that can be used as holistic indicators of soil quality.

Observational (e.g. earth observation, mapping, sensors) and analysis technologies (e.g. modelling, big data and analytics), combining observations with models, should be investigated to determine their potential to improve the implementation and monitoring of sustainable soil management in the UK.

Finally, the use of enzyme activity as an early indicator of soil structural degradation merits further investigation. For example, can the method detect differences in soil function earlier than the visual evaluation of soil structure or other measures of physical soil structure?
3.5 What is the practicality of different approaches?

Volume and characteristics of the overall evidence base

The search and screening process resulted in 16 research papers that specifically assessed the practicality of different approaches to assessing soil quality and ES delivery. Of the 16 papers identified, 15 investigated approaches related to food production, 10 related to water and nutrient cycling and ten to climate change mitigation (Table 6). Nine papers addressed physical indicators, six chemical indicators and eight biological indicators.

The papers covered visual evaluation of soil structure; the use of combinations of SQI’s as part of a screening process or indexing approach; soil microbial assessments; physical fractionation of organic matter; remote sensing techniques; ground penetrating radar and public participation as a means of increasing efficiency of soil quality monitoring, public engagement, and awareness of the importance of soil protection.

Table 6. The number of papers that investigate the practicality of different approaches in assessing soil quality and ES delivery.

<table>
<thead>
<tr>
<th>Indicator type</th>
<th>Ecosystem service</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Food production</td>
<td>Water and nutrient cycling</td>
</tr>
<tr>
<td>Physical</td>
<td>8</td>
<td>5</td>
</tr>
<tr>
<td>Chemical</td>
<td>6</td>
<td>2</td>
</tr>
<tr>
<td>Biological</td>
<td>7</td>
<td>6</td>
</tr>
<tr>
<td>Total</td>
<td>15</td>
<td>10</td>
</tr>
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</table>

What does the evidence base indicate in relation to the question posed?

Replication

The need for replication to provide meaningful results is a key consideration for assessing the suitability of SQI’s. Lark et al. (2014) described a methodology to develop robust protocols under geostatistical models to determine the number of samples needed to achieve a root mean square error (RMSE) of less than 5% of the sample mean value for both bulk density (BD) and volumetric SOC (topsoil and subsoil). The results indicate the relatively high number of samples needed to achieve a small (5%) RMSE (Figure 4); 16 points for a mineral soil and a peat soil, but on the peat soil at depths below 50 cm, the criterion could not be achieved with less than 25 cores. They conclude that more sites should be investigated to develop robust protocols for national-scale monitoring.
Visual evaluation

Cui et al. (2014) found that the Visual Evaluation of Soil Structure (VESS) was a suitable and practical method for assessing soil structural quality for grassland productivity. Furthermore, they measured significant negative correlations between the VESS soil quality score (Sq) and soil C content ($P<0.01$) and between Sq and soil N content ($P<0.05$), indicating a close relationship between soil structure and soil C and N.

SQI combinations and Indexing approaches

Bone et al. (2014) proposed a strategic set of measured SQI's for use in screening locations to assess the likelihood of soil degradation and indicate areas for further detailed assessment to address objections about the cost of soil protection policies. The paper also discusses and reviews methods and practicalities for data collection and screening, including the need for further pilot testing and protocol development. Use of public data collection could also allow more resource efficient protection of soils, in addition to benefits of public engagement, and raising awareness of the importance of soil protection.

Askari and Holden (2014) presented an indexing approach using a combination of physical and chemical properties (SOC, total N, aggregate size distribution, bulk density, bulk density of < 2 mm fraction, extractable potassium and C:N ratio) and concluded that it was a practical, time and cost effective method for quantitative evaluation of soil quality for grassland production under temperate maritime grassland management. Visual evaluation of soil structure was used to validate the indices.
Soil microbial assessments
Chapman et al. (2007) reviewed the various methods available for Community Level Physiological Profiles (CLPPs) assessment with respect to assessing the functional diversity of the soil microbial community. They found that a CLPP method known as MicroReSp (TM) (a multiple C-source, substrate induced respiration method using a “microtitre plate” format), offered a convenient, rapid and sensitive method for the determination of microbial functional diversity. They acknowledge that the “full potential for characterizing soil activity is yet to be realized”, but that the method could help improve understanding of how soil degradation pressures impact on basic soil functioning.

Creamer at al. (2009) assessed a number of practicality criteria for two soil microbiological techniques hitherto considered to have potential for use in soil monitoring schemes to detect changes in soil quality. They assessed the consistency, repeatability, intrinsic variation, inter-laboratory repeatability and land use discrimination of multi-enzyme activity assay and multiple substrate-induced respiration (MSIR). Discrimination of soils under different land uses was effective and relatively consistent, but intrinsic variation was large for both assays. Furthermore, the results produced by different laboratories were inconsistent, highlighting the need for the development of standard operating procedures including calibration and adequate replication. The results indicated that the two microbiological techniques were not suitable for use in soil quality monitoring schemes in their existing state of development.

Physical fractionation and remote sensing techniques
With particular focus on the analysis of SOM, Branco de Freitas Maia et al. (2013) reviewed the organic matter pools in soils and their functions. They argued that physical fractionation procedures can be used to isolate these pools and are highly relevant to assessing soil functionality, but are rarely adopted. They discussed the merits and demerits of wet oxidation procedures, relative to dry combustion for determining SOC contents and made reference to the emerging chemometric techniques based on the use of Near (NIR) and Mid (MIR) infrared spectroscopy. The use of physical fractionation procedures was also explored in Defra project SP0310 (To develop a robust indicator of soil organic matter status).

Defra project SP1316C assessed the opportunities provided by developments in earth observation and remote sensing to improve national scale monitoring of soil quality. It was concluded that remote sensing methods could be used in cultivated areas only to improve estimates of status and to enhance mapping, but not to monitor change, for a number of soil indicators such as extractable potassium (K), SOC, total N and soil pH. It may also be possible to use InSAR (interferometric synthetic aperture radar) satellite data to detect changes in the surface elevation of peat and cultivated soils to monitor peat depth and soil erosion respectively, using new processing techniques that produce elevation values sensitive to 1 mm over areas as small as one hundred square metres (Archer et al., 2014). Defra project SP1106 (sub-project IV: Exploration of methodologies for accurate routine determination of soil carbon) also investigated basic remote sensing methods for determining SOC.

Askari et al. (2015) argued that application of visible (VIS) and near-infrared (NIR) spectroscopy for prediction of soil properties may offer a cost and time effective approach for evaluation of soil structural quality. In a study of 20 arable and 20 grassland soils, soil
spectral reflectance produced acceptable models for predicting relevant soil structural indicators. They concluded that a combination of spectroscopic and chemometric techniques can be applied as a practical, rapid, low cost and quantitative approach for evaluating soil structural quality under arable and grassland management systems in Ireland. The method merits further investigation and development to help quantify soil structural condition and relate it to ES delivery.

Yang et al. (2012) assessed the potential for calibrating visible and near infrared (vis-NIR) spectra with total nitrogen (N), total carbon (C), organic C and inorganic C in soil on a 15-ha farm, using progressively fewer sets of wavelengths in the calibration with a view to developing a simpler and more cost effective spectrophotometry method for prediction of the above soil properties. Although these findings were only valid at the farm scale, the predictive models calibrated for the 100-nm interval spectra (21 wavelengths) performed almost as well as those for the 1-nm interval spectra (2100 wavelengths). Yang et al. (2012) recommended that the proposed wavelength reduction algorithms should be tested on a wider variety of soil types to assess the potential for its use in regional or national soil quality monitoring programmes.

Guerrero et al. (2014) found that the use of ‘spiked calibrations’ can minimize the efforts needed to use near-infrared (NIR) spectroscopy effectively for SOC assessment at local scales. However, the method is probably not robust at regional and national scales.

*Ground penetrating radar for peat depth estimation*

Fyfe et al. (2014) used ground penetrating radar (GPR) to estimate peat depths in a 4x1km survey area of blanket peat on Dartmoor, UK. Comparison of GPR estimates of peat depth correlated very well with core depths and provided the basis for a detailed understanding of the distribution of peat depths within the survey area. The results indicate a significant and previously unaccounted store of C within blanket peat regions which should be included in future calculations of overall C storage.

By contrast, Parry et al. (2014) found that GPR depth estimations, calibrated using common midpoint (CMP) surveys, were found to be on average 35% greater than probe measurements. However, they too suggested that GPR calibrated at each site using CMP surveys may provide a more accurate method for measuring peat depth than manual probing. Remote sensing methods are also being developed to use surface elevation of peat soils (and overlying vegetation) as a proxy for peat depth (Archer et al., 2014).

*Public participation*

Bone et al. (2012) investigate the potential use of public participation in soil monitoring and soil surveys. The Open Air Laboratories (OPAL) Soil and Earthworm Survey is given as an example of public participation in soil surveys to generate data that could be used to prioritise soil assessment and to address some of the concerns and objections to soil protection policies. They argued that such participative surveys could improve communication between scientists, regulators, stakeholders, and the wider society, and therefore increase the level of understanding and commitment to soil protection policies.
Danielsen et al. (2005) examined the potential of locally-based approaches to soil quality monitoring and covered the following key issues: cost, sustainability, their ability to detect true local or larger-scale trends, their links to management decisions and action, and the empowerment of local stakeholders. They believed that locally-based approaches could be more sustainable than many professional schemes, particularly when they are institutionalised within existing management structures, and linked to the delivery of ecosystem goods or services to local communities. When properly designed, local schemes can yield locally relevant results that can be as reliable as those derived from professional monitoring and can result in prompt action and behavioural change due to the higher level of engagement with local communities. However, more effort is needed to develop effective protocols for feeding locally-derived data up to national and international scales.

3.5.1 Implications for policy and further research

The evidence suggests that the development of a strategic set of measured SQI’s and the potential use of public participation in the measurement of certain suitable SQI’s merits further investigation.

For national-scale, scientific, soil monitoring networks, the findings indicate that more sites should be investigated to develop robust SQI measurement protocols. This would require intensive measurements to be carried out on arable, grassland, upland and lowland sites to cover the main agricultural land uses, agro-climatic zones and soil types in the UK. The following methods also merit further investigation and development:

- Remote sensing and the use of GPR as indicators of soil quality and to provide an accurate method for measuring peat depth in soil C accounting.

- Visible (VIS) and near-infrared (NIR) spectroscopy for prediction of soil properties and soil structural quality to help quantify soil structural condition and relate it to ES delivery.

The use of visual evaluation methods and spectroscopy or image processing methods should be incorporated into field-based experiments on soil management and long-term sites to improve understanding of the relationship between these measures of soil structural quality and to provide typical (reference) values under different soil type, land use and soil management combinations.

Standard operating procedures including calibration and guidance on adequate replication need to be developed for new soil microbiological techniques such as multi-enzyme activity assay and multiple substrate-induced respiration (MSIR).

3.6 Are there any established envelopes of normality for SQI’s?

Volume and characteristics of the overall evidence base

The search and screening process resulted in 14 research papers that provided key information on envelopes of normality for SQI’s. Of the 14 papers identified, six investigated envelopes of normality for SQI’s in relation to food production, twelve in relation to water and nutrient cycling and one to climate change mitigation (Table 7). Six papers provided
information on physical indicators, six on chemical indicators and eight on biological indicators.

Table 7. The number of papers that investigated envelopes of normality for SQI’s.

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<td>Total</td>
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<td>14</td>
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What does the evidence base indicate in relation to the question posed?

There is a significant volume of data providing typical values for indicators in the literature. The difficulty is interpreting this data with respect to ecosystem service delivery. There are a host of soil property datasets that can be used to provide statistics on typical values for indicators. In England and Wales, the following soil datasets are available although there are licensing issues that restrict access to some datasets (Defra project SP08003):

- LandIS - includes the National Soil Map (NATMAP), soil series attribute and function values (SOILSERIES), the National Soils Inventory (NSI), the Hydrology of Soil Types (HOST) and other soil data sets
- Soilscape (data available from Cranfield University; maps available on MAGIC – see below)
- Multi-Agency Geographic Information for the Countryside (MAGIC) - the government website providing geographic information on rural, urban, coastal and marine environments across Great Britain
- Representative Soil Sampling Scheme (RSSS)
- Countryside Survey (CS)
- Environment Change Network (ECN)
- G-BASE
- Agricultural Land Classification (ALC)
- Spatial Environmental Information System for Modelling the Impact of Chemicals (SEISMIC)
- National Soils Inventory (NSI)
- Hydrology of Soil Types (HOST)
- Forestry Commission Soil Classification (FCSC)

In Scotland, the following additional Scottish datasets are available:

- SSD - Scottish Soils Database held by the James Hutton Institute (part of NSI)
- LCA - Land Capability for Agriculture, held by the James Hutton Institute
- LCF - Land Capability for Forestry
- HMSD - Heavy Metals Sensitivities Database
• SSFIS - Scottish Soil Fertility Information Service, held by SRUC (Scotland’s Rural College)
• SLC - Scottish Land Cover (vegetation) database, held by the James Hutton Institute
• POP - Persistent Organic Pollutants database, held by the Scottish Environment Protection Agency (SEPA)
• RSE - Risk of Soil Erosion Maps (derived dataset based on Scottish Soils Database, part of NSI, held by the James Hutton Institute)
• CLD - Critical Loads Database, held by the Centre for Ecology and Hydrology (CEH)
• WMLR - Waste Management Licensing Regulations (database of soil analytical results from enforcement actions, held by SEPA)
• SSR - Sewage Sludge Regulations (database of soil analytical results from enforcement actions, held by SEPA)
• ESC - Ecological site classification
• SBBI - Scottish Blanket Bog Inventory
• NISAD - Soil Attributes Database Northern Ireland
• GLASOD - UNEP and International Soil Reference and Information Centre world map, the status of human-induced soil degradation
• LIDAR 2000 – remote sensed vegetation and land cover data
• SEPA project (LQ09) – constructed a catalogue of existing soil monitoring schemes

The UK Soil Indicators Consortium reviewed available data collected since the mid-1980s from the Soil Survey and Land Research soils database (i.e. LandIS data) and used this to create ‘trigger values’ or workable ranges related to concern and action levels for soil properties such as soil pH and bulk density (Merrington et al., 2006). Some trigger values were relatively straightforward to define, such as soil pH, but for other properties workable ranges depended on soil type and land use factors. For example, for bulk density it was
found that typical values were largely driven by SOM content rather than soil texture (Figure 5).

Physical and chemical indicators
Glendell et al. (2014) investigated issues of scale and spatial variability of soil properties under different land uses. They used a high resolution geostatistical approach to characterise the spatial variability of parameters, including soil bulk density (BD), total soil carbon (TC), nitrogen (TN), phosphorus (TP), inorganic phosphorus (IP), organic phosphorus (OP), stable N isotope ratio (delta N-15), C:N ratio, C storage and N storage in two study catchments with contrasting land uses (agricultural and semi-natural) that were subject to targeted management interventions to reduce flood risk and improve water quality. They found a stronger degree of spatial dependence of all soil properties in the agricultural than the semi-natural catchment except for bulk density and delta N-15. Furthermore, bulk density, TP, IP, OP, C:N ratio, delta N-15 and C storage showed a longer range or spatial auto-correlation in the agricultural catchment. They also found that while the national soil survey dataset can be relied upon for the broad characterisation of C and TP stocks, it underestimates the spatial variability of key soil properties in certain soil types and land uses.

Defra project SP0310 (To develop a robust indicator of soil organic matter status) produced statistically-derived “manageable” ranges of SOC for soils under different land uses (principally arable and permanent grassland) or “physiotopes”, i.e. environmental situation, largely driven by average annual precipitation. After excluding sites susceptible to flooding and soils with calcareous surface horizons or high pH, around 25% of total variation in SOC between sites was accounted for by a combination of soil clay content and average annual precipitation. For example, in the drier ‘physiotope’ the SOC range for sandy soils (< 10% clay) was 0.5 to 1.6%, while in the wetter clay ‘physiotope’ the corresponding range was 2.0 to 5.4%.

The visual evaluation of soil structure (VESS) method was developed to provide a quick, simple and easily understood test to enable researchers, farmers and consultants to score soil structural quality from a visual evaluation. Guimaraes et al. (2013) used the VESS method (soil quality - S-q - score) and measured related soil physical properties on soils of contrasting texture from native forest and from tracked and non-tracked arable soils after harvest. They tested the validity of S-q results compared with other measures of soil physical quality. S-q increased with penetration resistance and bulk density, but decreased with air permeability. The VESS method detected soil compaction even where it was not visible at the surface and as such may prove useful in diagnosing and remediating compaction (Figures 6 and 7).
Newell Price et al. (2013) reported on a survey of grassland soil compaction in England and Wales, in which soil visual evaluation methods were used alongside more widely accepted physical measurements of soil compaction (e.g. bulk density - BD and penetration resistance). Soil structural condition was investigated in 300 fields located on 150 farms, using the visual soil assessment (VSA) method from New Zealand and the Peerlkamp (soil structure - 'St') method (Figure 8).
Figure 8. Percentage of 300 grassland fields in England and Wales in each soil quality assessment category with 95% confidence intervals in brackets (source: Newell Price et al., 2013).

Jones et al. (2014) undertook a national large scale survey of the quality and quantity of DOC in 702 soils across Great Britain to evaluate its potential for evaluating changes in soil quality for water cycling and regulation in national soil quality monitoring programmes (Figure 9). They concluded that the quality of DOC (SUVA, total soluble phenolics) rather than its quantity provides a more useful measure of soil quality for water provision in large scale surveys.

Munro et al. (2002) assessed differences in SOM content, total N content, pH, topsoil depth, shear strength, and extractable phosphorus, potassium and magnesium between 14 paired conventionally and organically managed agricultural topsoils (28 fields). Organically managed topsoils were found to be characterized by deeper plough horizons and higher percentage organic matter, total N and extractable P. They had lower shear strengths and were substantially lower in extractable K than their conventional counterparts. The differences in soil physical and chemical properties most probably reflected contrasting management practices in terms of cultivation (deeper plough layers due to the need to incorporate organic manures) and the use of organic manures (higher SOC, total N and extractable P and lower shear strength) and manufactured fertiliser. However, there was limited information on the amount of organic manure applied to the organically and conventionally managed topsoils. Organic manure inputs are a significant driver for soil physico-chemical and biological properties (Bhogal et al., 2009).

Bhogal et al. (2009) measured the effects of organic carbon (OC) additions from farm manures and crop residues on selected soil bio-physical and physico-chemical properties at seven experimental sites, on contrasting soil types. The derived data provided key information on the amount and type of organic manure addition needed to produce a detectable change in key soil properties and improvements in soil quality. For example, total SOC increased by an average of 3% for every 10 t ha⁻¹ manure organic carbon (OC) applied, whereas light fraction SOC increased by c. 14%. The manure OC inputs (but not crop residue OC inputs) increased topsoil porosity by 0.6% and plant available water capacity by 2.5%, and decreased bulk density by 0.5% for every 10 t ha⁻¹ manure OC applied.
Envelopes of normality for many soil physico-chemical properties are well established (Merrington et al., 2006). These can help provide baseline values against which any changes to soil quality indicators can be compared in the future. However, there are a number of new emerging methods for measuring physical indicators such as aggregate stability (Rawlins et al., 2015) for which typical reference values and ranges need to be established for different combinations of soil type, land use and management. These values also need to be related to soil processes, functions and ES delivery. If this is not achieved we simply have an arbitrary scale of soil quality values that do not relate to ecosystem performance or what society needs from soil.
Biological indicators – macro-fauna

Smith et al. (2005) critically reviewed developments in risk assessment for terrestrial ecosystems and land contamination in the UK, with emphasis on deriving a measure of whether or not ecosystem function is altered as a result of land contamination. They proposed the use of earthworms as a favoured indicator species for protecting ecological function and provide evidence for relationships between earthworm number and levels of soil contamination, thereby helping to establish envelopes of normality for key indicators under varying conditions of contaminant stress.

Biological indicators – meso-fauna

Griffiths et al. (2012) used molecular characterisation of the nematode community to measure the effects of tillage intensity. Interestingly, they found that year and season were more important in affecting total nematode abundance, nematode channel ratio and proportion of fungal feeders than cultivation method. However, the effects of tillage were more clearly expressed when other bioindicators such as SOC content and fungal biomass were taken into account. More information is needed on the effects of management practices on the nematode community in a wider range of soil types and land uses. Nematode assemblages are sensitive to soil pollutants and because of their operation at multiple trophic levels are good indicators of the condition of the bacterial, fungal and protozoan soil communities (Pulleman et al., 2012).

Sousa et al. (2006) reported on changes in Collembola richness and diversity along a gradient of land use intensity from natural forest to agriculture in eight European countries (Portugal, Spain, France, Switzerland, Hungary, UK, Ireland and Finland). The total number of species per land use unit (LUU) was generally higher in natural forests and mixed-used landscapes, and lower in agricultural dominated landscapes. In forested areas, average species richness decreased along the gradient, showing that forest patches on mixed-use landscapes support a lower richness than in landscapes dominated by forest. Although a diverse landscape can support a high biodiversity, the results suggest that intensive fragmentation of the landscape can impact on local species richness with consequent possible implications for ES delivery at a regional scale.

Biological indicators – micro-fauna

Girvan et al. (2005) measured total soil bacterial community structure and microbial genetic diversity in two contrasting soils; an organo-mineral/improved pasture soil and a mineral/arable soil. The genetic diversity of the more diverse organo-mineral soil was more resistant to benzene perturbation than the less diverse mineral soil. Benzene amendment resulted in impairment of narrow niche function (as measured by mineralization of C-14 labelled 2,4-dichlorophenol) for both soils, but the organo-mineral soil recovered this function by the end of the experiment whereas the mineral soil did not. Copper treatment resulted in a large reduction in bacterial numbers and biomass, but there were only small differences in bacterial community diversity after nine weeks. The overall community structure was little altered and functionality, measured by mineralization rates, remained unchanged. The data indicated a degree of genetic and functional resistance to copper perturbation in soils with higher SOM content and/or genetic diversity, despite a significant reduction in bacterial numbers and biomass.

40
Smith *et al.* (2003) described the soil microbial community and fertility changes in a 10-year trial on mesotrophic grassland that was previously agriculturally improved. Management treatments were three hay-cut dates, plus two mineral fertilizer, two seed addition and two farmyard manure (FYM) applications. There were few treatment effects on the soil microflora. Bacterial biomass was reduced when FYM was applied with the 14 June cut date, but increased when FYM was applied with the 1 September cut date. The addition of seed, including legumes, and the absence of FYM was associated with an increase in soil fungi and an abundance of fungi relative to bacteria, indicating a functional role for individual microbial species and providing useful information on the nature of the microbial community under contrasting management regimes.

Robinson-Boyer *et al.* (2009) reported on the development of a methodology for the direct quantification of arbuscular mycorrhizal fungi (AMF) within roots. Quantitative polymerase chain reaction (PCR) provides the means to study spatial distribution and individual quantification of AMF in mixed communities over time. They concluded that the molecular tools now exist to quantitatively analyse the effect of environment, management or inoculation of soils on AMF communities within roots. It will therefore be possible to build up a picture of typical levels of AMF in soils under contrasting management regimes and the implications of this for nutrient cycling, particularly the potential of AMF to improve the availability of phosphorus in soils with low P reserves.

**Biological indicators – multiple**

Defra project SP0534 investigated the ability of a number of biological indicators to detect change in soil quality in UK soils (section 3.2). The project provided envelopes of normality in three different land use, soil type and management situations for a number of biological indicators, including Phospholipid fatty acid analysis (PLFAs); Terminal restriction fragment length polymorphism (TRFLP); multiple substrate induced respiration (MSIR); microarthropod diversity and biomass; and multi-enzymes. The quality, processes and functional capacity of urban soils can also provide additional evidence of the range of normality for soil properties. Hartley *et al.* (2008) investigated the use of biological rather than physico-chemical indicators to assess the soil quality of anthropogenic urban soils. Plant, invertebrate and microbial assays and functional processes were evaluated at 10 brownfield locations at different stages of remediation/contamination in northwest England. Extreme sites were discriminated on the basis of earthworm counts and a small number of indicators related to their activity. The results provided useful information on typical ranges for biological indicators, but they concluded that identifying a universally-applicable benchmark suite of biological indicators is very unlikely without considerable advancement of knowledge and technology.

Keith *et al.* (2012) reported on baseline data from a national survey in Ireland of soil physico-chemical properties and the abundance and diversity of micro-organisms (bacteria, fungi, mycorrhiza), and micro-, meso- and macro-fauna (nematodes; mites; earthworms, ants). They explored relationships between diversity and composition of the bioindicators across a general gradient representing dominant land uses (arable, pasture, rough-grazing, forest and bogland). Differences in diversity and composition of meso- and macro-fauna, but not microbes, were clear between agriculturally-managed (arable and pasture) and extensively-
managed (rough-grazing and “bogland”) soils corresponding to a broad division between 'mineral' and 'organic' soils. They suggested that different sets of bioindicators may be needed to assess soil quality under agricultural and extensive land use.

The EU project ‘Ecofinders’ aimed to standardise methodologies for the assessment of biological indicators and produce normal ranges for soil biodiversity indicators according to climatic zones, soil and land use types (Lemanceau, 2011). However, there will be an ongoing need to develop ‘normal’ ranges for new and emerging indicators at national and European scales.

3.6.1 Implications for policy and further research

Further work is needed on the standardisation of indicators. This has been started through national initiatives and European research projects. One such project was European Framework Programme ENVASSO (Environmental Assessment of Soil Monitoring), which was a first attempt to develop a harmonised system for soil quality and soil biodiversity monitoring across Europe. Procedures and protocols based on ISO standards were tested in pilot sites in France, Ireland, Portugal and Hungary to assess the efficiency and sensitivity of the indicators across European land use categories.

To interpret the results from such projects it will be important to define reference values to account for different combinations of soil type, land use and climate. These do not exist for biological indicators at a national or European scale although some density ranges have been published for different groups of organisms covering a selection of land use types in the Netherlands and France.

3.7 What do changes in soil biodiversity and soil C content mean for ecosystem service delivery?

Volume and characteristics of the overall evidence base

The search and screening process resulted in 17 research papers that investigated how changes in soil biodiversity and SOC content can influence ES delivery. Of the 17 papers identified, six investigated changes in relation to food production, eleven in relation to water and nutrient cycling and eight to climate change mitigation (Table 8). One paper included information on physical indicators, four on chemical indicators and 13 on biological indicators.

Table 8. The number of papers that investigate what changes in soil biodiversity and soil C content may mean for ecosystem service delivery.

<table>
<thead>
<tr>
<th>Indicator type</th>
<th>Ecosystem service</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Food production</td>
<td>Water and nutrient cycling</td>
</tr>
<tr>
<td>Physical</td>
<td>-</td>
<td>1</td>
</tr>
<tr>
<td>Chemical</td>
<td>1</td>
<td>2</td>
</tr>
<tr>
<td>Biological</td>
<td>5</td>
<td>10</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>6</strong></td>
<td><strong>11</strong></td>
</tr>
</tbody>
</table>
What does the evidence base indicate in relation to the question posed?

**Food production**

Loveland and Webb (2003) found little quantitative evidence that reduction in SOC concentrations in the soils of England and Wales will have marked effects on other soil properties or crop yields. However, they did suggest that more research was needed on the nature of SOC, particularly of the 'light' or 'fresh' fraction and its influence on soil properties under different land uses.

By contrast, Banwart et al. (2015) stated that maintaining and increasing SOC content can yield substantial multiple benefits. Greater SOC concentrations help to maintain soil structure and porosity providing greater air permeability and drainage for root growth and water holding capacity to sustain evapotranspiration and for this reason, policies are essential that encourage protecting, maintaining and enhancing soil C levels. Nziguheba et al. (2015) also highlighted the need to manage SOC to optimise a mix of five essential services — the provisioning of food, energy and water, regulating of climate and maintaining biodiversity.

Hooker et al. (1995) argued that AMF are essential to the sustainability of agricultural systems and that management practices such as minimum tillage, greater crop rotation, and integrated nutrient and pest management should be used to maximize their benefits. They stated that future research was needed to better understand the functional ecology of AMF in agroecosystems. However, they did not reflect on how AMF can be maintained within UK arable systems in which soil disturbance is a regular and often necessary part of soil management practice.

The EU project SOILSERVICE brought together natural scientists and economists in a transdisciplinary approach to understand how competition between land uses influences soil biodiversity and the provision of ecosystem goods and services. The SOILSERVICE project team developed an ecological-economic valuation tool that predicted that farm profits increase with an increase in SOC due to higher and more consistent crop yields (Breure et al., 2012). Farming systems seeking to maximise profits should therefore aim to adopt management practices that maintain or enhance SOC levels.

**Water and nutrient cycling**

A number of papers relate the soil microbial community to the provision of nutrient cycling services. For example, Deacon et al. (2006) investigated the functional role of saprotrophic soil fungi in the decomposition of organic matter in an upland grassland soil. They found that 'infrequent' microfungi were potentially more active in decomposition than the 'frequent' taxa, i.e. several had a higher overall activity, were able to utilise a wider range of substrates and were more combative than the 'abundant' taxa. This has implications for the importance of fungal diversity in C and nutrient cycling.

De Deyn et al. (2009) investigated interactions between plant species and soil microbes and found that changes in plant species and functional group richness influenced the cycling of both C and N in model grassland communities, but that these responses were mainly related to the presence and biomass of certain plant species, notably N fixers and forbs (broad-leaf,
herbaceous – not woody - flowering plants that are not graminoids; grasses, sedges and rushes).

Legay et al. (2014) investigated how soil abiotic properties and above- and below-ground plant traits affected N turnover at three mountain grassland sites in Europe. The results indicated that consideration of plant traits, and especially below-ground traits, increases the ability to predict the functional characteristics of microbial communities in grassland soils, such as the gene abundance of nitrifying and denitrifying microbes.

Everwand et al. (2014) investigated the potential for traits such as “community abundance-weighted means (CWM) of plant functional traits” and “measures of trait variability within a community (FDvar)” to predict C and water fluxes in different seasons. Their results indicated that database derived trait measures could improve the prediction of ecosystem C and water fluxes, but more detailed flux measurements were needed for validation.

There is also evidence that the amount and nature of SOC fractions can influence nutrient cycling in agricultural soils. For example, Murphy et al. (2011) used a 5-year old, replicated agricultural systems trial in a semi-arid environment with continuous wheat, crop rotation, crop-pasture rotation, annual pasture, and perennial pasture treatments to investigate: (i) whether agricultural systems which have increased plant residue inputs increase the amount of labile SOM relative to total SOM; and (ii) whether the size or quality of OM fractions is most strongly linked to the size, activity, functional diversity, and community structure of the soil microbial biomass. The size, function, and structure of the soil microbial community were better correlated with SOM fractions than total SOC; and the C:N ratio of light fraction organic matter was related to the amount of potentially mineralisable N in soil. Overall, their findings supported the premise that labile fractions of SOM are more strongly related to microbial community structure and function than is total SOM.

Climate change mitigation
Buckingham, et al. (2014) collated information on Scottish soil C stocks and C losses and reviewed the potential pressures on terrestrial C, future C stocks and the related delivery of ES. They suggested that limited long-term data indicated little evidence of significant changes in the C stocks of Scottish soils, and stressed the importance of more detailed monitoring of soil C stocks for different land management practice and land use changes to provide further insight into the potential changes in sequestered C.

Beaumont et al. (2014) provided information about the capacity of coastal habitats in the UK to sequester and store CO₂ and highlighted global issues regarding the quantification and valuation of C sequestration and storage. They stress that while our ability to value ecosystem services is improving, considerable uncertainty remains.

Some researchers have found an apparent relationship between plant diversity and the ability to sequester soil C. For example, Fornara and Tilman (2008) investigated the long-term effects of plant functional diversity on rates of soil C accumulation in N-limited grasslands at Cedar Creek, Minnesota, USA. High-diversity mixtures of perennial grassland plant species stored five and six-times more soil C and N, on average, than monoculture plots of the same species. Furthermore, the presence of C4 grasses and legumes increased
soil C accumulation by two and five-times respectively. They argue that the introduction of key C4 grass-legume species, and associated soil microbial diversity, could increase soil C accumulation and biomass production in both high- and low-diversity N-limited grassland systems.

De Deyn et al. (2011) investigated whether or not biodiversity restoration in grassland soils had additional benefits for C and N storage. They found that seeding with a plant diversity mix of herbs and legumes increased soil C and N storage especially when it included red clover (Trifolium pratense), sequestering 0.32 t C and 350 kg N ha\(^{-1}\) year\(^{-1}\) in the most successful management treatment. Carbon and nitrogen accumulation was associated with reduced ecosystem respiration, increased SOM content and improved soil structure. However, withholding the application of manufactured fertiliser reduced the amount of C and N in vegetation, which can have implications for livestock nutrition.

There is also some evidence that the diversity and abundance of certain bacterial species can have implications for C fluxes in peatlands. For example, Fenner et al. (2005) used temporal temperature gradient gel electrophoresis (TTGGE) to assess the effect of simulated summer drought and increased rainfall on the diversity of phenolic degrading bacteria in a northern peatland. Under simulated drought, a greater diversity (130%, \(P < 0.05\)) and abundance of phenolic catabolising bacterial species was found, resulting in lower concentrations of phenolic compounds, DOC and increased CO\(_2\) fluxes. The increased abundance and diversity of phenolic catabolising bacteria was therefore associated with increased mineralisation, a lower carbon storage capacity and increased climate ‘forcing’. By contrast, simulated increased rainfall reduced the diversity and abundance of phenolic degrading bacteria and increased DOC concentrations and anaerobic trace gas fluxes. Bacterial diversity was therefore linked to DOC concentrations and the colour of water draining northern peatlands.

**Multiple benefits**

Loveland and Webb (2003) discussed the implications of loss of SOM for the productive capacity of agriculture and soil nutrient cycling mechanisms in the UK. They reviewed what is known about critical thresholds of SOC or SOM, mainly in soils of temperate regions, and concluded that the quantitative evidence for such thresholds is slight and needs significant development, although there was some evidence that there may be a desirable range of SOC for a wide spectrum of soils.

Bradford et al. (2014) investigated the role that biodiversity plays in delivering five ecosystem services related to plant productivity, C storage, and nutrient turnover. They manipulated soil faunal community composition of model grassland ecosystems in combination with the use of manufactured fertiliser N and found that the functional complexity of the soil communities had a consistent positive effect on multifunctionality indices. However, these indices were not effective in advancing theoretical understanding of how biodiversity is related to the provision of multiple ecosystem services. Only two of the five ecosystem processes responded positively to increasing complexity. Indeed, Bradford et al. (2014) concluded that the use of such indices can obscure relationships that exist between communities and key ecosystem processes.
De Vries et al. (2013) quantified how differences in soil food web composition resulting from land use systems (intensive wheat rotation, extensive rotation, and permanent grassland) influenced soil functions and ES delivery across four European countries with contrasting soils and climate. ‘Intensive’ wheat rotation reduced food web biomass; soil food web properties were a better predictor of C and N cycling processes than land use; earthworm biomass and fungal/bacterial energy channel ratio were correlated with both land use and C cycling processes; and N cycling was best explained by food web characteristics such as AMF and bacterial channel biomass rather than land use. De Vries et al. (2013) conclude that soil biota should be included in C and N cycling models, highlighting the need to conserve soil biodiversity for C and nutrient cycling.

Fornara et al. (2009) reported that plant functional composition on a grassland soil affected fine root mass loss, root detritus N dynamics and net soil N mineralization rates through effects on root chemistry. However, there was no evidence of increased root or soil N immobilization rates with increased below-ground plant biomass (i.e. increased soil C inputs). They concluded that the simultaneous presence of different plant functional groups (in plant mixtures) with opposite effects on root mass loss, root N release and soil N mineralization rates may be crucial for sustaining multiple ecosystem services such as food production, climate change mitigation and C sequestration in many N-limited grassland systems.

Lavelle et al. (1997) reviewed the interactions between plant, animal and microbial components of the soil biota and argued that earthworms are the most important ecosystem engineers in European terrestrial ecosystems. They presented evidence that earthworms may influence the diversity and activity of soil biota in subordinate trophic levels such as decomposers, nutrient transformers and ‘biocontrollers’ (i.e. small invertebrates that act as herbivores or predate on other invertebrates or micro-organisms; Pulleman et al., 2012). Lavelle et al. (1997) also described the links between the activity and diversity of earthworms (different ecotypes) and aspects of soil physical morphology and function such as structural heterogeneity, aggregate stability, organic matter distribution and water regulation.

One approach for linking soil degradation, soil biodiversity and ecosystem services is based on ecological traits, that is “the morphological, physiological, behavioural or life-history attributes of organisms” (Pulleman et al., 2012). Lavorel et al. (2013) presented a conceptual framework for applying trait-based approaches to predict the impact of environmental change on ecosystem service delivery across multiple trophic levels. They provide worked examples of functional relationships and trophic interactions to demonstrate the flexibility and value of the framework for linking disparate data sources, identifying knowledge gaps and generating hypotheses for quantitative models.

3.7.1 Implications for policy and further research

The evidence highlights the importance of protecting SOC and conserving soil biodiversity for multiple benefits, including sustainable (profitable) food production, C and nutrient cycling and climate change mitigation. Policies that encourage protecting, maintaining and enhancing soil C levels will be essential in the future to optimise the provisioning of food, energy and water, regulating of climate and maintaining biodiversity.
Further research is needed to better understand:

- The nature of SOC, particularly the 'light' or 'fresh' fraction and its influence on soil properties under different land uses.
- The implications of manufactured fertiliser use for C sequestration and storage and the balance between climate change mitigation considerations and food production.
- The key factors and processes influencing C fluxes in peatlands.
- The functional ecology of AMF and other soil biota in different agricultural systems and possible synergies and antagonisms with a need for sustainable intensification.
- The importance of fungal diversity in C and nutrient cycling.
- How soil biodiversity is related to the provision of multiple ecosystem services.
- How to value the ES provided by SOC and soil biodiversity.

There is also a need for:

- More detailed monitoring of soil C stocks for different land use, management practices and land use changes to provide further insight into the potential changes in sequestered C.
- More quantitative evidence to develop critical thresholds and workable ranges for SOC for different soil types, land uses and ecosystem services.
- More detailed measurements of C, nutrient and water fluxes to better understand the relationship between ecological traits and ES delivery.
4. Soil degradation

4.1 Introduction
This section focuses on the principal soil degradation processes in the UK; namely, compaction, loss of organic matter, erosion and acidification. While many studies have investigated soil degradation processes, fewer have explicitly quantified the impacts on soils to delivery ecosystem services.

Thirteen research papers addressed the question of how degradation processes significantly affect soil quality and function (Figure 10). Eleven papers investigated how soil degradation is best measured and if any tipping point for soil function and the delivery of key ecosystem services can be identified; and two papers addressed how climate change is likely to affect soil degradation processes.

Figure 10. Number of papers identified as relevant to addressing specific questions within the “soil degradation” theme.

4.2 At what point do degradation processes significantly affect soil quality and function?

Volume and characteristics of the overall evidence base

The search and screening process resulted in 21 research papers that addressed the point at which degradation processes significantly impact on soil quality and function. However, these 21 papers also referenced many other relevant documents and sources. Of the 21 papers identified one explicitly covered food production, ten water and nutrient cycling and one climate change mitigation (Table 9). In some cases, the ecosystem services that the paper addressed was not specified in the keywords or abstract.
Table 9. The number of papers that address how soil degradation affects specific ecosystem service delivery by specific degradation processes (loss of soil organic matter, erosion and acidification).

<table>
<thead>
<tr>
<th>Ecosystem service</th>
<th>Degradation process</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Food production</td>
</tr>
<tr>
<td>Loss of organic matter</td>
<td>1</td>
</tr>
<tr>
<td>Erosion</td>
<td>-</td>
</tr>
<tr>
<td>Acidification</td>
<td>-</td>
</tr>
<tr>
<td>Total</td>
<td>1</td>
</tr>
</tbody>
</table>

Few papers specified the soil types that were addressed (Table 10). The papers that focused on the impacts of erosion, were largely conducted on mineral soils.

Table 10. The number of papers that investigate the impact of soil degradation on ecosystem service delivery in mineral, organo-mineral and peaty soils specifically.

<table>
<thead>
<tr>
<th>Soil type</th>
<th>Mineral</th>
<th>Organo-mineral</th>
<th>Peaty</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>3</td>
<td>-</td>
<td>2</td>
</tr>
</tbody>
</table>

*What does the evidence base indicate in relation to the question posed?*

*Soil Erosion*

Accelerated erosion by processes of water, wind and crop harvesting represent a serious threat to soils and the delivery of ecosystem services (Bilotta *et al*., 2012). It is known that human activity such as cultivation will increase the rate of soil erosion. Consequently, the majority of studies identified, have focused upon quantifying soil erosion rates or testing mitigation strategies on agricultural soils. In a recent meta-analysis García-Ruiz *et al.* (2015) highlighted the significance of soil erosion from agricultural soils. It was found that whilst erosion rates are extremely variable across the world, there was a significant effect of land use, with arable soils yielding the highest erosion rates, and forest and shrubland yielding the lowest.

In order to help evaluate the impact that soil erosion will have on soil function the concept/definition of ‘tolerable soil erosion’ has been developed by many researchers. Verheijen *et al.* (2014), provided a comprehensive review of the definition of tolerable soil erosion rates (Table 11). Typically, tolerable soil erosion definitions can be placed into one of two broad themes:

i. Tolerable soil erosion maintains the dynamic equilibrium of soil quantity; and

ii. A tolerable soil erosion rate does not impact negatively on the biomass productivity of the soil.
Table 11. Interpretations and definitions for ‘tolerable soil erosion’ taken from Verheijen et al. (2009).

<table>
<thead>
<tr>
<th>Definitions/ interpretation</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>The maximum volume of erosion-removed topsoil that provides high, or economically feasible, fertility for a long time</td>
<td>Patsukevich et al. (1997)</td>
</tr>
<tr>
<td>Soil loss balanced by soil formation through weathering of rocks</td>
<td>Roose (1996)</td>
</tr>
<tr>
<td>Erosion that does not lead to any appreciable reduction in soil productivity</td>
<td>Roose (1996)</td>
</tr>
<tr>
<td>The maximum rate of soil erosion that permits an optimum level of crop productivity to be sustained economically and indefinitely</td>
<td>ISSS (1996)</td>
</tr>
<tr>
<td>The average annual soil loss a given soil type may experience and still maintain its productivity over an extended period of time (permissible soil loss)</td>
<td>Kok et al. (1995)</td>
</tr>
<tr>
<td>The maximum permissible rate of erosion at which soil fertility can be maintained over 20-25 years</td>
<td>Morgan (2005)</td>
</tr>
<tr>
<td>i) The maximum average annual soil loss that will allow continuous cropping and maintain soil productivity without requiring additional management inputs.</td>
<td>SSSA (2001)</td>
</tr>
<tr>
<td>ii) The maximum soil erosion loss that is offset by the theoretical maximum rate of soil development which will maintain an equilibrium between soil losses and gains</td>
<td>SSSA (2001)</td>
</tr>
<tr>
<td>Rate of soil erosion is not larger than the rate of soil production (acceptable rates of soil erosion)</td>
<td>Boardman and Poesen (2006)</td>
</tr>
</tbody>
</table>

Verheijen et al. (2014), suggest that the impact of soil erosion on soil function should be widened to include ecosystem services other than biomass production. Proposing that the impact of soil erosion should be evaluated against 4 primary functions: 1) protection of habitats, 2) cultural/historical information, 3) production, 4) engineering and 5) regulation of resources, Verheijen et al. (2009) define tolerable soil erosion rates as: “any mean annual soil erosion rate at which a deterioration or loss of one or more soil functions does not occur”. In comparison, the actual soil erosion rate is defined as: “the total amount of soil lost by all recognised soil erosion processes”. However, Verheijen et al. (2009) explain that there is still a degree of ambiguity with this definition; judgements have to be made at what stage soil function is considered to be reduced and at what scale this should be assessed.

Verheijen et al. (2009) explain that, defined baseline and threshold values are required for the evaluation of soil monitoring data – to achieve this accurate data on both soil loss and genesis is required. Verheijen et al. (2009) define the upper limit of tolerable soil erosion rates, as equal to soil formation at c.1.4 t ha⁻¹ yr⁻¹ whereas the lower limit is c.0.3 t ha⁻¹ yr⁻¹. However, it was concluded that reported rates of actual soil erosion rates across Europe were 3 to 40 times greater than the upper limit of tolerable soil erosion, with substantial
spatio-temporal variation. This is comparable to the rates reported by Pimentel (2006), who reported that actual rates of soil erosion were 10-40 times greater than tolerable limits. Verheijen et al. (2009) found that soil erosion only exceeds tolerable rates when the soil was cultivated or affected by other human disturbances. It was noted that substantial effort is required to reduce soil erosion rates to tolerable levels; this is particularly true for tilled arable soils.

Bakker et al. (2007) conducted an analysis of available data from experiments investigating the impact of soil erosion on agricultural production, in order to assess if a relationship could be detected between crop yields and soil water availability to plants which was defined as the most important yield-determining factor affected by soil erosion. An overall finding from the study was that yield reductions at the field scale were around 4% for each 0.1 m of soil lost. The yield reductions were attributed to a reduction in rooting depth and or plant available water (as reported by e.g. Olson et al., 1999; Baker et al., 2004).

Bakker et al. (2007) mapped the spatial distribution of erosion rates and calculated the potential for erosion-induced productivity losses. It was concluded that future reductions in productivity in Europe, particularly northern Europe will be small. However, Bakker et al. (2007) noted that the negative impacts of soil erosion on the sustainability of agricultural practices should also be considered. Firstly, to maintain production increasing inputs will be required to compensate for nutrient losses. Secondly, the offsite effects of fertiliser, pesticide and herbicide losses on vulnerable terrestrial and aquatic ecosystems should also be considered.

Some studies have investigated the impacts of soil erosion on wider ecosystem services. Lewis et al. (2013) explained that the weed seedbank is central to the biodiversity in temperate agro-ecosystems of northern Europe. Lewis et al. (2013) assessed the likely impact of soil erosion on the composition and abundance of the arable weed seedbank, by taking into account both the erosion mechanisms affecting arable land and factors influencing arable weed seedbank abundance. Using a mean net erosion rate of \(c.7 \, \text{t} \, \text{ha}^{-1} \, \text{yr}^{-1}\) and seedbank densities of \(c.2000 \, \text{seeds} \, \text{m}^{-2}\) the average annual loss of seed inventory is \(c.0.5\% \, \text{yr}^{-1}\). Over a period of 20 years average soil erosion rates could export \(c.10\%\) of the arable weed seedbank.

Artz et al. (2013) reported that net abatement benefits from peatland restoration could provide up to \(9 \, \text{t} \, \text{CO}_2\text{e} \, \text{ha}^{-1} \, \text{yr}^{-1}\), however the actual savings will vary between sites and with the extent of degradation. Furthermore, early intervention on less damaged bogs will prevent the habitats from becoming severely degraded and more highly emitting. Lilly et al. (Sniffer project UKCC21) concluded that, the risk of erosion of organic and organo-mineral soils within Scotland and Northern Ireland is greatest on actively eroding peatland. Areas most at-risk were identified as, those which are grazed by sheep and have moderate to high numbers of red deer (>15 deer per square kilometre).

A number of studies have investigated the effectiveness of mitigation strategies to reduce soil erosion. Rickson (2014) reviewed the scientific evidence of the efficacy of land management practices to reduce soil erosion and sediment losses. In total, 73 mitigation strategies were identified; however, empirical data are limited and the effectiveness of
measures vary with both time and location. It is known that the effectiveness of measures will vary depending on a number of factors including: rainfall, timing of the intervention, crop type, soil type, slope, drainage and vegetation cover. Given the limitations of the data in terms of geographical coverage, duration of monitoring the effectiveness of mitigation measures cannot be extrapolated to other regions.

In terms of impacts on policy, Rickson (2014) concluded that the uncertainty in the effectiveness of mitigation measures has implications for ensuring that stipulated environmental targets are met. For instance, for water courses to meet ‘good ecological status’ (GES), mitigation measures will need to reduce sediment loss from diffuse agricultural sources by 20% (Collins & Anthony, 2008b; Collins et al., 2009). However, in catchments failing Water Framework Directive (WFD) standards the effectiveness of measures would have to increase substantially by as much as 80% and mitigation measures suitable for use in modern farming systems will not meet this target. Furthermore, uncertainty over the effectiveness of soil erosion mitigation measures may not encourage uptake by farmers.

Broadman and Favis-Morlock (2014) explained that while early sowing of autumn cereals is advised this may not be possible if conditions are wet and crucial factors affecting this include the onset of autumn rainfall and the need for ‘stale seedbeds’ (sequences of cultivation to control weeds). The study developed a conceptual model of soil erosion by water erosion, focusing on the relationship between developing crop cover and the timing of rainfall. It was found that the timing of winter rainfall cannot be predicted, therefore, erosion advice to farmers based on choice of drilling date is difficult to formulate and likely to be ineffective. It was advised that “better thought out” mitigation strategies were required.

A number of studies evaluated the risk of surface runoff and soil erosion from different agricultural land uses, by assessing the extent of soil structural degradation features. For example, Boardman et al. (2009) conducted an assessment in West Sussex, UK, to identify which fields were at most risk from erosion and most likely to cause damage to roads and rivers due to muddy runoff. It was found that an increased risk of surface runoff was associated with certain crops: potatoes, winter cereals, maize and grazed turnips. It was concluded that flood management strategies should control enhanced surface runoff (and consequent flooding) at the source; one approach would be to include appropriate land management as a flood management solution.

Holman et al. (2003), examined soils prone to structural damage in England and Wales under five common lowland cropping systems: autumn-sown crops, late-harvested crops, field vegetables, orchards and sheep fattening/livestock rearing systems. It was demonstrated that soil structural degradation was prevalent over a wider range of soil types and land uses than previously reported, with soil structural damage widespread across all five land uses and a range of soil types. ‘Severe’ soil structural damage was reported for late-harvested crops such as maize, sugar beet, maincrop potatoes and autumn-sown crops.
Acidification

Overall, few of the studies identified directly investigated the impacts of acidification on the ability of soils to deliver ecosystem services. The studies identified focused upon addressing the effects of N deposition on nutrient poor semi-natural grassland and heathland, through increasing nutrient supply and acidification; mainly by assessing the effects upon vegetative species diversity.

Dupre et al. (2010), investigated the negative impact of N deposition on species richness in acidic grasslands, using vegetation data collected over a period of c.70 years. It was found that species richness of vascular plants and bryophytes decreased with time and was affected by various factors, soil pH, latitude and N deposition. It was found that N deposition explained more of the variation in species richness. This finding supports the theory that declining species richness is caused by increasing N availability and less so by changes in management practices or soil acidification.

Power et al. (2006) monitored the recovery of a lowland heathland in southern England, after 7 years of N inputs and assessed changes in vegetation growth, soil chemistry and the soil microbial community. It was found that parameters (e.g. soil pH) recovered to pre-treatment levels rapidly whereas as others (e.g. vegetation cover and microbial activity) respond much more slowly. It was concluded that the ecological effects of small amounts of N deposition will persist for many years, possibly several decades after N deposition is reduced.

Van den Berg et al. (2011), analysed permanent quadrat data from nature reserves on calcareous grassland sites in the UK, to determine the effects of N deposition on species composition change. The results showed that N deposition had no significant spatial association with species richness, diversity or frequency of species adapted to low N-conditions. However temporal analysis showed a significant association with changes in Shannon diversity and evenness. In summary, a decline in rare and scarce species was found despite management practices in place to maintain high biodiversity and characteristic species. This was most likely due to both the direct effects of N deposition and indirect effects due to changes in soil pH. It was concluded that, even management strategies aiming to maintain high biodiversity cannot prevent the negative impacts of N deposition.

Studies investigating the impacts of N deposition on C loss from peat bogs include: Bragazza et al. (2006), Jones et al. (2008) and Evans et al. (2012) (Section 4.4.) and Rowe et al. (2014)

Bragazza et al. (2006), reported that atmospheric N deposition promoted C loss from peat bogs, this was quantified by increases in carbon dioxide emissions and DOC release. It was concluded that the increased losses of C was a consequence of increased N availability which favoured microbial decomposition. Indeed, the ongoing effects of nitrogen deposition may offset the benefits of lower sulphur deposition (Emmett et al., 2007).

In a more recent study Bragazaa et al. (2012) investigated the effect of long-term N-deposition on the decomposition of bog plant litter. Overall it was found that N-deposition interacts with both above and below ground biodiversity. It was found that 7-years of N addition decreased the C: N ratio of litter, increased bacterial biomass and the activity of
hydrolytic and oxidative enzymes and led to a shift in the balance between peat-forming mosses and vascular plants. Jones et al. (2008) tested the hypothesis that peatland C and N cycles are regulated by acidity change and that rising pH will increase the loss of C and N as dissolved organic matter (DOM) due to both increases in biological productivity and increasing DOM solubility. This hypothesis was further developed by the ‘MADOC’ model (Rowe et al., 2014), which simulated that DOC increases due to N-pollution will become evident and sustained after soil pH has stabilised.

4.2.1 Implications for policy and future research

Soil erosion

The evidence indicates that soil erosion only exceeds tolerable rates when the soil is cultivated or affected by other human disturbances (including overgrazing by livestock). In some circumstances, substantial effort is required to reduce soil erosion rates to tolerable levels; particularly for tilled sandy and light silty soils on sloping land. This has implications for policy in terms of the degree to which soil erosion needs to be controlled to maintain soil functions and soil natural capital and the types of soil management practices that need to be encouraged in particular circumstances. The uncertainty in the effectiveness of mitigation measures has implications for ensuring that stipulated environmental targets are met. The on-going Defra project SP1318 (Scaling up the benefits of field scale soil protection measures to understand their impact at the landscape scale) aims to address some of these issues.

García-Ruiz et al. (2015) summarised opinions on priorities for research proposed by other researchers. Their findings and the evidence provided above indicate that further research is required to:

- Provide additional information on both baseline soil erosion rates and the efficacy of mitigation strategies through further field-based monitoring across a wide range of locations (Evans, 2000; Toy et al., 2002).
- Improve the modelling of “soil erosion rates and sediment yield at regional scales under present and future climate and land use scenarios” that takes account of the typical magnitude and variability in observed soil erosion rates (Vente et al., 2013).
- Develop numerical models of soil formation to help implement soil erosion mitigation strategies at appropriate spatial scales (Verheijen et al., 2009).

4.3 How is soil degradation best measured and can any tipping points for soil function and the delivery of the key ecosystem services be identified?

Volume and characteristics of the overall evidence base

The search and screening process resulted in eight research papers that addressed the point at which degradation processes significantly impact on soil quality and function. However, these eight papers also referenced many other relevant documents and sources. None of the papers identified explicitly addressed food production or climate change mitigation (Table 12). In some cases, the ecosystem services that the paper addressed was not specified in the keywords or abstract.
Table 12. The number of papers that addressed how soil degradation is best measured and whether any tipping points for soil function and the delivery of the key ecosystem services can be identified; broken down by specific degradation processes (loss of soil organic matter, erosion and acidification).

<table>
<thead>
<tr>
<th>Degradation process</th>
<th>Food production</th>
<th>Water and nutrient cycling</th>
<th>Climate change mitigation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Loss of organic matter</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Erosion</td>
<td>-</td>
<td>4</td>
<td>-</td>
</tr>
<tr>
<td>Acidification</td>
<td>-</td>
<td>3</td>
<td>-</td>
</tr>
<tr>
<td>Total</td>
<td>-</td>
<td>7</td>
<td>-</td>
</tr>
</tbody>
</table>

Few papers specified which soil types were addressed (Table 13). The papers that focused on the impacts of erosion, were largely conducted on mineral soils.

Table 13. The number of papers that investigate the impact of soil degradation on ecosystem service delivery in mineral, organo-mineral and peaty soils specifically.

<table>
<thead>
<tr>
<th></th>
<th>Mineral</th>
<th>Organo-mineral</th>
<th>Peaty</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>4</td>
<td>-</td>
<td>1</td>
</tr>
</tbody>
</table>

What does the evidence base indicate in relation to the question posed?

Measuring soil degradation

The aspirational 2011 Scottish Environmental Monitoring Strategy developed by the coordinated Agenda for Marine Environmental and Rural Affairs Science (CAMERAS) provides a framework for monitoring, and encourages co-ordination of monitoring programmes within Scotland. As part of this programme a soil Monitoring Action Plan (MAP) has been proposed (http://www.environment.scotland.gov.uk/media/59999/Soil_Monitoring_Action_Plan.PDF) as there is a clear need for soil data not only for soil protection but also a wider range of policy, management and planning issues. It was also highlighted that while historical information on Scottish soils is available there is a lack of information on how Scottish soils are changing and if any change is leading to soil degradation. Black et al., (2013) identified a wide range of soil functions and issues (e.g. erosion, compaction, SOC, climate change) that should be addressed by a monitoring programme.

McBratney et al. (2014) described the use of ‘reference states’ to determine whether soil degradation has occurred. For example, Droogers and Bouma (1997) proposed that soil should be classified according to its genoform and phenoform. The genoform would be a soil in its natural or ‘reference’ state, recognising what we know about soil and its genesis (e.g. Jenny, 1941). The phenoform takes account of how a soil has been altered, including the effects of soil management. Examples include, lack of vegetation cover resulting in erosion and loss of topsoil depth; or decades of organic material addition resulting in higher soil organic matter content (Bouma, 2002). McBratney et al. (2014) state that the genoform would usually be assumed to be the reference state for a soil, but that there is a point at
which soil alteration may be so significant (Milà i Canals et al., 2007) that a phenoform may become the reference state, for example following soil degradation or organic matter addition.

**Soil erosion**

Many of the review papers discussed different methods and scales for measuring soil erosion. For example, Verheijen et al. (2009) provided a comprehensive summary of the different spatial temporal scales over which different soil erosion processes operate and explained that connecting these different scales (*i.e.* **scaling measurements up or down**) is a current major challenge for research. Rates of soil erosion can be determined using several methods with significant variation between methods in terms of cost, practicability and reliability: 1) plot & field measurements, 2) soil erosion modelling, 3) mass/energy balance modelling 4) radionuclide measurement, 5) suspended sediment loads in rivers and streams. Rickson et al. (2014), concluded that information on soil erosion following the implementation of mitigation measures are available at the field scale but are highly variable over space and time. The effectiveness of measures to reduce soil erosion losses is dependent upon a number of factors including: rainfall, crop type, soil type, drainage, vegetative cover, cultivation techniques and the timing of operations. Therefore, the effectiveness of mitigation strategies should not be extrapolated to different land uses, soil types or spatial scales (e.g. field to farm to catchment).

Soil erosion indicators proposed by Eckelmann et al. (2006) included: 1) estimated soil loss by water via rill, inter-rill and sheet erosion 2) estimated soil loss by wind erosion and 3) estimated loss by tillage erosion.

Identifying priority areas for soil protection has been proposed (European Commission, 2006). Kibblewhite et al. (2014), as part of Defra project SP1609 (“Exploring the Priority Area Approach”), developed a method for estimating risk of harm to soil e.g. due to soil erosion (by water and wind) and organic matter decline with the aim of defining risk-based priority areas for soil protection (*i.e.* **areas where the balance of cost and benefits of risk reduction measures is expected to be more favourable**). Overall it was concluded that there are technical limitations to both the definition and implementation of soil protection priority areas.

Sniffer project (DP0W2) concluded that Citizen Science could be used to: 1) identify and record occurrences of erosion, 2) identify possible causes and drivers of erosion; 3) monitor where erosion occurs; 4) identify shifts or changes over time; and 5) validate risk-based modelling approaches. However, in order to validate process based models, Citizen Science data would have to be complemented with repeated measurements of eroded sediments.

Young et al. (2015), reported that wind abrasion resistance (WAR) of Machair soils were strongly correlated with: soil organic matter concentration, persistence of water repellence, arsenic concentration, mean particle size, water content at time of sampling and sand mineralogy. Furthermore, regression tree analysis was capable of predicting resistance to wind erosion by placing soils into broad WAR classes *i.e.* high (WAR >35), moderate (>15 to <35) and low (<15). Further work is required to assess the seasonal and annual variability of
WAR across Machair grassland habitats and to devise appropriate methods for increasing soil WAR while not impacting on the unique biodiversity of the Machair habitats.

McHugh (2000) assessed upland erosion in England and Wales across 399 field sites. Short-term erosion rates were determined through cross-sectional traverses on erosion gullies in 1997 and 1999. While longer-term variation were assessed by interpreting aerial photographs taken between 1946 and 1989. Overall, the results showed that upland eroded areas in England and Wales increased by over 518 ha between 1997 and 1999. The increase in upland erosion was largely attributed (99.9%) to humans (e.g. amenity and agricultural use) and livestock (e.g. grazing causing, scars, tracks and poaching).

Acidification

In an ammonia (NH3) fumigation study Prendergast-Miller et al. (2008), investigated the response of enchytraeid worms to NH3-N deposition in an ombrotrophic bog. It was found that neither litter quality, worm abundance nor diversity was affected by NH3-N deposition. This was despite increases in peat pH and mineral N. It was concluded that enchytraeids were not sensitive indicators of NH3-N deposition on ombrotrophic bogs that have a low critical N-loading.

N-deposition

Black et al. (2010) (SEPA report HP801) identified soil indicators to assess the impact of atmospheric N-deposition from point sources on soil quality in habitats of conservation interest in Scotland. A set of indicators was proposed that would inform on the 5 soil functions set out within the Scottish Soil Framework. The indicators included: soil C:N ratio; fungal species fruiting bodies; bacterial to fungal ratio; base cation: aluminium (Al) ratio; soil pH soil solution NH4; NO3 (concentration and proportion); and phosphomonoesterase. Black et al. (2010) highlighted that there was a requirement as part of the monitoring programme to establish baseline or reference values and that existing soils data could be used to develop these.

Tipping points

Soil erosion

The study by Verheihen et al. (2012) has been widely referenced as providing a tolerable or threshold rate of soil erosion. Verheihen et al. (2012), suggested that tolerable or threshold rates of soil erosion in Europe can be set equal to estimated soil formation rates, i.e. 1 t ha⁻¹ yr⁻¹. However, further research is required to define how this threshold will impact on soil functions. Virto et al. (2015) highlighted that erosion rates in countries across Western Europe typically exceed the tolerable rate of 1 t ha⁻¹ yr⁻¹. However the impacts of exceeding this threshold on specific soil functions is not defined, therefore it appears that the tolerable rate of 1 t ha⁻¹ yr⁻¹ cannot be defined as a ‘tipping point’ for soil function. Furthermore, Rickson (2014) explained that, even when there is a consensus on what targets of soil erosion should be achieved, consistently achieving these targets using mitigation strategies is unlikely given the variability in the efficacy indicated by existing data.
**Organic matter**

The issue of thresholds for SOC was reviewed by Loveland and Webb (2003). They concluded that the quantitative evidence for thresholds of SOC is slight. However, it was highlighted that there is some evidence there might be a desirable range of SOC covering a wide spectrum of soils, although further quantitative evidence for this is needed. The review also assessed the impact of a reduction in SOC concentration on crops yields stating that reductions could have a marked effect on other soil properties and crop yields.

More recently, Verheijen et al. (2005) explained that critical or threshold levels of SOC in relation to soil physical properties and nutrient supply to crops has been researched extensively (Greenland et al. 1975; Watts & Dexter 1997; Six et al. 2000; Chenu et al. 2000; Carter 2002; Shepherd et al. 2002). It was explained that studies have been unable to identify critical or threshold levels of SOC and this is most likely to be a reflection of the many possible ways that SOC interacts with the soil, water and plant system. Furthermore, there is large spatial and temporal variation in SOC concentration and composition due to differences in soil properties, climate and land use etc.

Verheijen et al. (2005) reported that two environmental variables, clay content and precipitation, had a strong influence on %SOC and together explained 25% of the variation in the SOC concentration of arable and ley-arable soils in England and Wales. Indicative SOC management ranges were proposed; the %SOC in dry-sandy soil ranged from 0.5 to 1.6%, while in wet-clayey soil it was from 2.0 to 5.4%. However, it was stipulated that the proposed upper and lower limit should not be confused with critical or threshold SOC values that refer to a particular SOC level at which soil functions change significantly.

Eckelmann et al. (2006) reported that, the Technical Group on Organic Matter of the Soil Thematic Strategy (Van Camp et al. 2004c) state that only very general threshold values can be proposed for organic matter decline (Table 14), but that ideally regionally defined and validated thresholds should exist.

**Table 14. Preliminary approach to identify first Tier thresholds for SOC levels (source: Eckelmann et al., 2006)**

<table>
<thead>
<tr>
<th>Soil &lt; 2% SOC</th>
<th>Arable soils, in particular those that are managed in continuous arable production, especially where tillage is intensive</th>
</tr>
</thead>
<tbody>
<tr>
<td>Soil &gt; 8% SOC</td>
<td>Drained, current or formerly wet soils under arable crops or intensive livestock management</td>
</tr>
</tbody>
</table>

A number of researchers have proposed that 1% SOC is the critical level for soil quality decline, as this is the point at which water limited yield potential will not be reached and other key functions not performed (Lal, 2013; Kay and Angers, 1999 in Merante et al., 2014). However, it has been widely reported that no single critical SOC threshold can be proposed across soil types and climatic regions (Schjønning et al., 2009; Verheijen et al., 2005; van Camp et al., 2004 in Merante et al., 2014).
Peat soils have large water contents and a high porosity. However, they have a low saturated hydraulic conductivity and can store large amounts of water, which helps them to retain their saturated water status (Holden et al., 2007) and to reduce the rate of organic matter mineralisation. However, peat soils will also shrink and swell with changes in moisture (Schwärzel et al., 2002) and when artificially drained bio-oxidation of organic matter in peat soils can lead to irreversible changes in soil physical characteristics (Kechavarzi, et al., 2010). Kechavarzi, et al. (2010) reported how the hydraulic properties of peat are significantly influenced by the degree of degradation, with decomposition and humification resulting in the loss of structural pores, thereby reducing the soils ability to store, retain and transmit water.

Defra project SP0556 (‘A compendium of peat restoration and management projects’, 2008) compiled data on the success of restoration projects through questionnaires, interviews and survey of recent scientific work. The report highlighted that monitoring of restoration progress is essential to ensure that restoration work objectives are achieved including the provision of a data resource that can be analysed to inform best practice.

**Acidification**

Several studies have addressed critical loads of N deposition on nutrient poor semi-natural grassland and heathland habitats. For example, Bobbink et al. (2003 and 2010) proposed that critical loads for calcareous grasslands of N deposition, should be in the range 15-25 kg N ha\(^{-1}\) yr\(^{-1}\); and Van Den Berg et al. (2011) demonstrated that exceeding this critical load results in a decrease in species diversity and in the prevalence of characteristic species.

Power et al. (2006) applied 15 kg N ha\(^{-1}\) yr\(^{-1}\) over 7 years to a lowland heathland in England. It was found that, 7 years after the last N applications, impacts on nutrient cycling were still detected with elevated rates of microbial activity detected. The results implied that it may take many years for nutrient cycling to recover to former levels following a reduction in N-deposition.

Phoenix et al. (2012) reviewed the responses to N deposition in the UKREATE network – nine experimental heathland sites located across the UK covering grassland, bog and dune ecosystems. The study reported increases in soil N%, (KCl) extractable N, N-cycling rates and acid-base status, in response to low N pollutant loadings of 7.7-10 kg ha\(^{-1}\) yr\(^{-1}\). Thus, relatively low N-inputs can have a detrimental impact on ecosystems due to accumulated loadings.

**N-deposition**

Black et al. (2010) concluded that a soil monitoring programme to assess the impact of atmospheric N-deposition on sensitive habitats would give the opportunity to validate published critical thresholds e.g. soil C:N, pH base cation: Al ratio. Further work is also required to develop thresholds for, soil solution NH\(_4\) and NO\(_3\) (concentration and proportion), bacterial to fungal ratio and phosphomonoesterase, in sensitive habitats to atmospheric N-deposition.
4.3.1 Implications for policy and future research

Few studies have quantified linkages and thresholds between the change in soil properties and associated changes in soil processes. Unless changes in soil properties can be linked to soil processes, it will be difficult to understand and predict the impact of soil degradation in a meaningful way. The SoilTrEC project ‘soil transformations in European Catchments’ (http://www.soiltrec.eu/index.html), in part addressed this issue. Critical Zone Observatories (CZOs) across a range of diverse habitats were established, allowing multi-disciplinary teams to investigate the processes of soil formation and degradation, with the ultimate aim of producing quantitative predictive models (Banwart, 2011).

Erosion
We do not currently understand how ‘tolerable’ rates of soil erosion (i.e. erosion rates that do not exceed the rate of soil formation) relate to ecosystem functioning. Therefore, further work is needed to identify particular ‘tipping points’ for soil function. Nevertheless, given limited resources, identifying priority areas for soil protection would be a practical and desirable step. However, Defra project SP1609 concluded that, there are technical limitations to both the definition and implementation of soil protection priority areas.

Organic matter
While some researchers have proposed upper and lower limits for SOC based on environmental conditions and soil type (e.g. Verheijen et al., 2005) – this is distinct from a ‘tipping point’ or critical point at which soil function is lost. Furthermore, many researchers have concluded that for SOC no single critical SOC threshold can be proposed across soil types and climatic regions. Nevertheless, losses of SOM have been associated with a loss in soil quality and function; and reduced resistance and resilience to perturbation. Policies that maintain or enhance SOM contents should therefore result in multiple benefits for society.

N-deposition
The evidence indicates that relatively low N-inputs can have a detrimental impact on ecosystems in terms of microbial activity and nutrient cycling due to accumulated loadings and that some habitats such as lowland heathland can take ten years or more to recover from elevated N deposition. Other strategies such as the targeted removal of upper soil horizons should therefore be tested to help habitats recover from nutrient enrichment.

To further our understanding of levels of soil degradation, the following research areas should be explored:

- More quantitative relationships should be developed between soil properties, attributes, processes and ecosystem services (Palm et al., 2007).
- In addition to defining ‘tipping points’ for soil function, research is required to understand the effects of incipient degradation (e.g. erosion and compaction) processes upon ecosystem functioning.
- Given the large spatial and temporal variation of soil erosion rates, more resources should be focused on field monitoring of soil erosion than on modelling (Verheijen et al., 2009; Trimble & Crosson, 2000; Brazier, 2004).
- More field research is required to determine the typical range of effectiveness values for mitigation strategies to help ecosystems recover from soil degradation.
• Further research is required to assess the impact of N deposition on species-rich grassland under different management strategy scenarios (Van den Berg et al., 2011).
• More field research is required to understand how rates of soil erosion, %SOC etc. impact on soil functions and the delivery of ecosystem services.
• More research on a wider range of environmental conditions is needed to further develop and differentiate the ‘SOC management range’ concept (Verheijen et al., 2005).
• To build on the SoilTrEC project and the establishment of the CZO’s, future areas of research should include more integration of: 1) international research methodologies and 2) government and commercial activity (Banwart, 2011).

4.4 How is climate change likely to affect soil degradation processes?

Volume and characteristics of the overall evidence base

Overall, the evidence identified by the literature search was very scarce, with just two papers directly addressing this question. However, Defra project SP0571 Modelling the impact of climate change on soils using UK Climate Projections) provided some useful indications of possible climate change impacts based on data from one member of the HadRM3 11-member climate projection ensemble. Other papers and sources known to contain relevant information and/or identified at the project workshops have also been included.

What does the evidence base indicate in relation to the question posed?

Soil erosion

Defra project SP0571 (Modelling the impact of climate change on soils using UK Climate Projections) used the PESERA (Pan-European Soil Erosion Risk Assessment) model to assess impacts of climate change projections on soil erosion. The results indicated a projected increase in (spatially averaged) erosion in general to the end of the century due to increased winter rainfall. Increases were predicted to be greatest in upland areas of England and Wales.

Mullan (2013) applied the Water Erosion Prediction Project (WEPP) model to 6 hillslopes in Northern Ireland to assess the statistical significance of future soil erosion rates based on relative changes to natural variability. Overall, the results indicated a mix of soil erosion rate increases and decreases, depending on which scenarios were considered. It was found that downscaled climate change projections resulted in erosion rate decreases, whereas under scenarios in which rainfall intensity and land use changes were accounted for, large increases were predicted.

Organic matter

Defra project SP0571 (Modelling the impact of climate change on soils using UK Climate Projections) used the HadRM3Q0 climate projection ensemble under a medium emission scenario and the ECOSSE (Estimating Carbon in Organic Soils - Sequestration and Emissions)
model to predict relatively small changes in soil C content in England and Wales over the period 2010 to 2080.

Two surveys have reported contrasting impacts of climate change on soil organic carbon (SOC):

Using the National Soil Inventory of England and Wales (NSI), Bellamy et al. (2003) attributed large declines in SOC from the period 1978-1983 and 1995-2003 to climate change. By contrast, using results from the Countryside Survey (CS) Carey et al. (2008), Emmett et al. (2010) and Norton et al. (2012) reported that, within the 1978-2007 period, there was no significant change in SOC concentration.

More recently, Barraclough et al. (2015), used regression models based on space-for-time substitution on the data from the NSI study (1978-1983), combined with changes in rainfall and air temperature over the same period, to determine how much of the changes in C stocks could be accounted for by changes in climate. Overall, it was found that most of the changes in SOC reported by Bellamy et al. (2005) were not consistent with changes in climate. It was also reported that different soil types would respond differently. For organo-mineral/mineral soils only 0-5% of the change in soil C can be predicted by climate. While, for organic soils, it is estimated that between 9-22% can be predicted by climate. It was concluded that organo-mineral/mineral and organic soils under temperate conditions will respond differently to changes in climate. Carbon concentration in mineral soils being weakly positively correlated with rainfall, but insensitive to temperature while C in organic soils is strongly negatively corrected with temperature.

From 1993 to 2007, Stutter et al. (2011) used long-term soil monitoring datasets to assess DOC losses from moorland C reserves in organo-mineral soils at three key UK Environmental Change Network sites. They found that soil solution DOC concentrations increased in both surface and subsoil horizons of a freely-draining podzol (48% and 215% increases in O and Bs horizons, respectively). By contrast, DOC concentrations declined in a gleyed podzol and there was no change in a peat soil. It was concluded that the effects of the key factors (ionic strength, acid deposition recovery, soil hydrology and temperature) on DOC solubility could not be separated, but that climate change was probably one of the factors influencing DOC losses.

Acidification

Soils in England and Wales are in the process of recovery from acidification in the recent past. Results from Defra project 0571 indicated that increased weathering under projected climate change would tend to accelerate recovery, while reduced runoff would slow down recovery. The combined effect was estimated to make little difference to recovery. However, only the temperature dependence of weathering was considered in the analysis. There may be other soil processes with temperature dependence that could contribute additional or confounding effects.

Evans et al. (2012), investigated DOC concentration in surface waters, which have increased across much of Europe and North America. Reasons for the increases in surface water DOC concentration, include: changing climate, land management, eutrophication and acid deposition. In a 4-year, four-site replicated field experiment located on two moorlands in
the UK, both acidifying and de-acidifying treatments were tested. The study reported a positive relationship between DOC and acidity. It was concluded that, changing soil acidity may have wider impacts on ecosystem C balances than changing climate and that decreasing sulphur deposition may be accelerating terrestrial C loss.

Implications for policy and further research

Barraclough et al. (2015) recommended that future monitoring of SOC stocks should focus on soils with soil C concentrations above 250 and 435 g kg\(^{-1}\).

Evans et al. (2012), recommend that further work is required “to establish the impacts of changing soil acidity on the wider C cycle and to ensure that observed changes in terrestrial C cycling, particularly those based on measurements in industrialised regions, are not erroneously attributed to other drivers.”

Monitoring of soil compaction (soil structure and bulk density) and soil erosion (soil depth and soil erosion features) will be important to determine whether any changes in the nature and timing of rainfall, and its relationship with soil management practices, is affecting the severity of these degradation processes.
5. Sustainable soils / Aspirational soil quality targets

5.1 Introduction

Soil sustainability can be defined as the continued ability of a soil to provide essential ecosystem services and soil functions, as well as its resilience to change. A sustainable soil is therefore one that is able to perform the key functions that society requires from it, both now and in the future, is resistant (does not change to a significant degree) and resilient (recovers to its former state) to disturbance and perturbation (Defra SP1605). However, within the context of climate change, some degree of change may have to be expected in the long term (many decades) if there are changes to soil and air temperatures and soil wetness regimes (Barraclough et al., 2015). McBratney et al. (2014) proposed that the key to sustainable soil use was to match its intended use to its capability i.e. “soil should not only be viewed through a lens focusing on its ability to produce”.

There has been much discussion on sustainable soil management as central to safeguarding soils (e.g. Natural Environment White Paper, 2011, Defra Evidence Plan, 2011, Scottish Soil Strategy, 2009), however, what are the properties of a sustainable soil, what should we be aspiring to achieve and what impact will climate change have on these aspirations? This fits with the concept of the “right” soil (Bouma, 2005), which considers how stakeholders deal with and require of soil in a policy-making context. Soil quality specifications have been produced for sports pitches, land restoration and forestry (e.g. Defra, 2005; Moffat, 2003; SNIFFER, 2010; Sport England, 2011), including soil physical and chemical indicators. However, such specifications are not formally applied to agricultural soils to the same extent. This section explores the evidence base for answering these questions, and evaluates whether we have the necessary models in order to understand how sustainable soils deliver key ecosystem services (notably, food production, water and nutrient cycling and climate change mitigation).

A total of 47 research papers addressed this overall subject area. 14 considered the properties of a sustainable soil; 9 considered targets for soil quality; 9 looked at models for soil sustainability and ecosystem service delivery; 15 considered the impact of climate change on soil quality targets; (Figure 11). There was also further work reported in a number of Defra projects (e.g. Defra SP1601, sub-project A on the delivery of soil functions/services and SP1605 sub-project D on what makes some soils more sustainable/resilient to change).

5.2 What properties should a sustainable soil have?

Volume and characteristics of the overall evidence base

The search and screening process resulted in 14 research papers on the topic of sustainable soils. A number were review papers that provided a conceptual framework for assessing various soil functions (e.g. Banwart et al., 2012), whilst others focused on specific soil properties e.g. soil moisture regime (Moyano et al., 2012), metal concentrations (Creamer et al., 2008) and soil-plant interactions (Eisenhauer et al., 2010). Almost half of the papers considered grassland soils and the plant/soil characteristics that were important for achieving high diversity, nutrient retention and C storage (as opposed to high yield). One paper looked specifically at peat soils and how respiration rates are affected by temperature (Chapman & Thurlow, 1988). A review by Mueller et al., (2010) specifically addressed the productivity function of both grass and arable soils, but there were no papers that focused
solely on what the key characteristics of a sustainably managed arable soil should be. In addition to the papers obtained from the literature search, Defra project SP1605, subproject D, provided useful insights into what makes soils resilient; Defra project SP0550 developed a mapping system to cover the multifunctional capacity of soils, based on the original Agricultural Land Classification (ALC) system; and Defra project SP1104 investigated the implications of UK Climate Projections (UKCP09) on the future distribution of ALC grades and the best and most versatile land in England and Wales.

What does the evidence base indicate in relation to the question posed?

Banwart et al. (2012) used the concept of ‘Critical Zones’ to describe soil processes and functions along a life cycle of soil development, with the functions delivered by a soil dependent on its position within that cycle: soil formation; development; productive use; loss of function & degradation. This provides a useful framework for assessing and characterising soils in terms of their ability to perform essential ecosystem services. Indeed, Dickie et al. (2013) looked specifically at changes in mycorrhizal communities as soils went through these phases of development/degradation, but observed no consistent shift in populations to suggest that a particular population structure was associated with a specific phase in a soils development. Eisenhauer et al. (2010) observed successional change in microbial characteristics of grassland soils from a disturbed (zymogeneous) to an established (autochthonous) microbial community four years after grassland establishment.

Figure 11. Number of papers identified as relevant to addressing specific questions within the “sustainable soils” theme.
Defra project SP1605 showed that soil type, parent material and texture were important in determining soil resilience – a key feature of soil sustainability, which shows how well a soil can recover following a disturbance. The nature of the soil microbial community was also seen to be important, with land use and surprisingly organic matter content having low significance (although indirectly organic matter was considered to have a role through its influence on soil biology). However, the characteristics of a sustainable soil are likely to be context-dependent, with different soils delivering some ecosystem services more effectively than others and therefore potentially having different criteria for sustainability (Chambers & Bhogal, 2012).

A minimum level of SOC is probably critical for soils to be able to deliver multiple benefits (Loveland and Webb, 2003; Van Noordwijk et al., 2015). However, although it would be very helpful for policy making to define quantitative targets on the basis of thresholds, at this stage there is probably not sufficient scientific evidence to define such thresholds beyond specific case studies and locations. Nevertheless, such reference sites and values could be very important as a framework for soil monitoring.

Robinson et al. (2014) state that in order to set goals for soil quality and ‘soil health’ there is a need to better define these terms (soil quality for what?). Concepts such as ‘soil health’ and ‘sustainable’ are difficult to legislate for because wanting improved soils depends on what improved is, for what use and on what time scale? It is often easier to specify the things we do not want to happen (i.e. soil degradation), than describe what the ideal or “right” soil should be. Robinson et al. (2014) state that “soil science needs to carefully consider better ways to set goals and objectives that can be used in policy and management development, and for valuation”.

Several papers considered grassland soils, particularly in relation to above ground diversity and the delivery of the services of C and N cycling. For example, De-Vries et al. (2011) tested the assumption that higher fungal biomass in grassland soils results in lower N losses. Their laboratory study revealed that nitrous oxide (N₂O) emissions and denitrification were indeed lower where grassland soils had a higher fungal biomass. Nitrate leaching losses were also almost halved on soils with a higher fungal biomass. Orwin et al. (2010) suggested using plant traits as a predictor of functioning and showed that grass species with high growth rate, produced high quality litter (i.e. easily decomposable) and occurred on soils dominated by bacterial communities which resulted in high rates of N mineralisation and higher soil extractable P, although surprisingly respiration rates were lower.

For peatlands, initiatives such as the ‘Moorlands for the Future Partnership’ have measured and recorded the effects of moorland restoration and management on soil and water quality (http://www.moorsforthefuture.org.uk/). This has allowed the impact of restoration to be assessed, conservation successes quantified and future targets for peatland quality identified (e.g. Moors for the Future, 2007). The project has contributed to an overview of peatland restoration and management activities across the UK through the peatland compendium (Defra project SP0556 - A compendium of UK peat restoration and management projects).
With regard to food production, a key question posed by Mueller et al. (2010) is very pertinent: “which properties of soil most affect their productivity?” On a global scale, they observed that the main constraints to soil productivity (i.e. food production) were soil moisture and thermal regime, followed by what they term ‘internal deficiencies’ i.e. shallow/stony soils that impede rooting or adverse chemistry such as acidity or salinity, with topography also considered to be important (in terms of impacts on erosion and accessibility). For example, a number of studies have demonstrated that low soil pH (i.e. soil acidity) can impact on crop productivity (e.g. Bolton 1977; Johnston and Whinam, 1980; Johnston, 2001). However at the local (field) scale, soil structure was seen as a ‘crucial criterion’ of agricultural soil quality, with ‘unfavourable’ structure resulting in lower crop yields, greater leaching/runoff and erosion. Indeed, they suggested that the preservation of soil structure was ‘key to sustaining soil function’ and the use of visual soil assessment methods (e.g. VSA, Shepherd, 2009) a powerful tool for guiding soil management decisions. Besides soil structure and from a broader perspective of soil functionality, the authors highlighted SOC as a key indicator of soil quality, associated with many soil functions other than productivity. They observed that the main soil classification schemes (including the world reference basis) provided little information on soil functionality and there was a need for a standard method for assessing soil productivity across a range of scales.

Soil moisture regime has also been suggested to be important for the function of soil C storage (Moyano et al., 2014) through its control on respiration and hence loss of C. Temperature is also important, particularly for peat soils (Chapman & Thurlow, 1998).

Only one paper dealt with the issue of soil contamination and its impact on soil functions. Creamer et al., (2008) observed that when topsoil total Zn, Cu and Ni concentrations were within the UK statutory limits, there was no effect on invertebrate abundance or diversity. However, at concentrations above the current limits, although there was little impact on overall diversity, some individual taxa were affected; earthworm populations were reduced at elevated Cu concentrations, while nematode and enchytraeid worms were sensitive to elevated concentrations of both Cu and Zn, with resultant lower rates of litter decomposition, and hence potential consequences for nutrient cycling.

By contrast, in Environment Agency project SC050054SR1, Nicholson et al. (2008) found that the existing statutory limit values in the Sludge (Use in Agriculture) Regulations and the Code of Practice for Agricultural Use of Sewage Sludge may not be sufficiently protective of soil quality in terms of crop yield; wheat grain cadmium or lead concentration; microbial biomass; respiration rate; and/or rhizobia, earthworm, nematode or enchytraeid numbers. However, the authors stated that there was no evidence to determine whether the observed reduction in biological indicator values were in fact ‘harmful’ and that further work was needed to establish the magnitude of change in a biological parameter that should be considered unacceptable. Overall, the project concluded that there was a balance to be struck between environmental protection and regulation and the sustainable recycling of organic material to land.
5.2.1 Implications for policy and further research

- There is a clear need to define different criteria for soil sustainability depending on the ecosystem service in question and to define concepts such as ‘soil health’ and ‘sustainable’ in order to set goals and objectives that can be used in policy and management development.

- More scientific evidence to define thresholds or critical levels for SOC and soil biodiversity for a wider range of soil type, agro-climatic and land use scenarios would also help guide policy and provide more information on soil status.

- The maintenance or enhancement of soil organic matter is a key driver for maintaining soil quality and ensuring the delivery of multiple soil functions (e.g. Gregory et al., 2009). However, research is also needed to bring together a range of physical, chemical and biological indicators to define sustainability for different soil types and the delivery of specific soil functions and ecosystem services.

- To maintain an integrated understanding of the effect of soil management on soil properties it will be important to sustain work on soil physics and soil engineering disciplines as well as soil chemistry and biology. Workshop delegates commented that there had been a reduced emphasis on soil physics and engineering within research remits in recent years (e.g. the BBSRC research framework).

- Given that the preservation of soil structure is “key to sustaining soil function” there is a continued need to develop rapid methods of soil structural assessment from practical visual evaluation methods that can be used in the field by practitioners to the development of new technology to provide research tools that are more effective at quantifying soil structure on a continuous scale. There is also a need for a standard method for assessing soil productivity across a range of scales, as identified by Mueller et al. (2010).

- There is also a continued need to investigate the implications of soil management practices, particularly the input of organic materials and manufactured fertilisers, for soil quality in general and soil metal concentrations in particular. High soil metal concentrations can compromise soil function, particularly the activity and abundance of N-fixing bacteria, and also have implications for livestock and human health. Regulations are useful for setting ‘safe’ limits for metals in soils, but if techniques are not found to reduce the annual addition of metals to some soils their capacity to attenuate potentially toxic elements could ultimately be exhausted.
5.3 What targets should we have for soil quality?

**Volume and characteristics of the overall evidence base**

Karlen *et al.* (1997) defined soil quality as “the capacity of a specific kind of soil to function within natural or managed ecosystem boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and support human health and habitation” and (more simply) “the capacity (of soil) to function”.

Doran and Zeiss (2000) proposed a similar definition for ‘soil condition’; that is “the capacity of a soil to function within land use and ecosystem boundaries, to sustain biological productivity, maintain environmental health, and promote plant, animal and human health. The condition of a soil can be inferred by measuring specific soil properties (e.g., organic matter content) and by observing soil status (e.g. fertility).”

A total of 18 research papers were considered to address the subject of targets for soil quality/condition. Over half of them (10 papers), focused on soil biology/biodiversity and its role in determining ecosystem functioning (e.g. Anderson, 2009; Bell *et al.*, 2005; Thiele-Bruhn *et al.*, 2012; & Usher *et al.*, 2006); three looked at the importance of plant/soil interactions and the influence of above-ground biomass on soils (A'Bear *et al.*, 2014; Grigulis *et al.*, 2013 & Jackson *et al.*, 2007); one discussion paper considered approaches to assessing soil quality (Bone *et al.*, 2010); one emphasised the importance of taking spatial variation in soil properties into account (Glendell *et al.*, 2014), with only one paper focusing specifically on peat soils and the importance of measuring/modelling depth of peat particularly for evaluating the ecosystem service of C storage and water regulation (Parry *et al.* 2012).

However, although many of the papers were useful in identifying what properties are important for soil quality and ecosystem service delivery, very few gave insight into what we should be aspiring to achieve, and if we can indeed set targets for soil quality. Only two papers made some attempt at this in relation to soil erosion and organic matter loss (Verheijen *et al.*, 2012; Kibblewhite *et al.*, 2014).

**What does the evidence base indicate in relation to the question posed?**

Over the last decade, the importance and role of soil biology and biodiversity for soil functioning and ecosystem service delivery has become increasingly studied as new techniques help open up what was once a ‘black box’ (Usher *et al.* 2006). Studies have demonstrated clear differences in the biodiversity and community structure of different land uses (e.g. arable vs. grassland, low input vs. high input systems; Theile Bruhn *et al.*, 2012; De Vries *et al.*, 2013; Grigulis *et al.*, 2013), and how these differences can impact on key ecosystem services. For example, systems more dominated by bacterial microbial communities have been linked to increased N losses and reduced C storage (e.g. Grigulis *et al.*, 2013). Likewise, aspects of the whole food web (micro-, macro- and meso-fauna) have been strongly correlated to C and N cycling processes across a variety of land uses (De-Vries *et al.*, 2013).

However, none of these studies provide insight into what we should be aspiring to achieve in terms of targets for soil biology and biodiversity, although a biologically diverse system is inferred. Not all species will be equally important providers of ecosystem services. Blouin *et
al., (2013) discuss the importance of earthworms as ‘ecosystem engineers’, contributing to the development of soil structure and nutrient cycling, thereby vital to the ecosystem services of food production, water and climate regulation. This would suggest that the presence of earthworms should be a key aspiration for soil quality. However, no targets have been proposed and Blouin et al. (2013) suggest further work is required on the merits and risks of earthworm introduction into fields. A’Bear et al. (2014) emphasised the importance of taking into account plant-soil interactions and demonstrated the influence of above-ground and below-ground herbivory, as well as mycorrhizal interactions in controlling mineralisation (and hence nutrient cycling), as well as pollination and biological control of pests.

Kibblewhite et al. (2008) proposed that the majority of soil ecosystem services were largely driven by biological processes, underpinned by SOC decomposition, and that quantifying the flow of C between functions is essential for the assessment of soil health. Kibblewhite et al. (2014) go on to explore methods for estimating the risk of harm to soil by soil organic matter decline and soil erosion. For mineral soils, the rate of change of topsoil SOC concentrations (g kg\(^{-1}\) yr\(^{-1}\)) was suggested as an indicator of potential harm, with the rate of change in soil C stocks (t ha\(^{-1}\) yr\(^{-1}\)) used for organic soils. A rate of no loss of SOC/C stock was suggested as being the most acceptable, given the size of the total SOC pool (e.g. a 0.3% loss in SOC would equate to an annual loss of 2.2 x 10\(^6\)t C from soils in England and Wales).

For erosion, Kibblewhite et al. (2014) built on the work of Verheijen et al. (2009) and their proposed ‘tolerable erosion threshold’ of 1 t ha\(^{-1}\) yr\(^{-1}\). Given the difficulty in establishing ‘critical levels of soil organic matter’ (Loveland and Webb, 2003) and ‘targets’ for other soil properties, looking at acceptable rates of change may be more appropriate. Glendell et al. (2014) also stressed that spatial variation in soil properties should also be taken into account when setting targets and evaluating the impact of land management on soil function and associated ecosystem services.

5.3.1 Implications for policy and further research

None of the studies identified set targets for soil biology and biodiversity. This is one area that merits further consideration within a soil type, land use, climate and ES framework. Deriving targets for soil quality that are context-dependent (i.e. dependent on the soil type, climate and land use) is likely to require a systems-based approach. Rather than setting absolute targets for soil quality across a range of systems, assessing the rate and direction of change in key soil properties is more appropriate. It will be necessary to define acceptable levels of change or thresholds, which will also be dependent on the soil function/ecosystem service being considered.

Priorities for further work include:

- The merits and risks of earthworm introduction into fields and other forms of soil biota/food web manipulation.
- Developing a methodology to quantify the risk of harm to soil by soil organic matter decline (Defra projects SP0306 – “Critical levels of soil organic matter” – and SP1606 – “The total costs of soil degradation in England and Wales”) and soil erosion (Defra project SP1317 – “How does a loss of soil depth impact on the ability to deliver vital ecosystem services”).
• Establishing acceptable rates of change in key soil properties such as bulk density, SOC and functional groups of soil biota for different combinations of soil type, land use, climate and ES.

5.4 Is there a model for sustainable soils to deliver the key ecosystem services?

Volume and characteristics of the overall evidence base

The search and screening process resulted in nine research papers that provide information that relate to the development of a model or framework for sustainable soils (Table 15). Of the nine papers identified, five papers described models, frameworks or the use of critical zone observatories to help provide guidance on sustainable soils (Aitkenhead et al., 2011; Banwart et al., 2011; Banwart et al., 2012; O’Sullivan & Simota, 1995; Wilkinson et al., 2014). One paper looked at working towards sustainable land management (Cowie et al., 2011) and another on the role of sediment in sustaining ecosystem services (Apitz, 2012). Only three papers focused on particular ecosystem services or indicator types; two papers focused on biological indicators for nutrient cycling (A’Bear et al., 2014; De Deyn et al., 2009); and one focused on modelling the environmental impacts of compaction (O’Sullivan & Simota, 1995).

Table 15. The number of papers that provide information for developing a model for sustainable soils by ecosystem service and indicator type.

<table>
<thead>
<tr>
<th>Indicator type</th>
<th>Ecosystem service</th>
<th>Food production</th>
<th>Water and nutrient cycling</th>
<th>Climate change mitigation</th>
<th>Total</th>
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<td></td>
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<td>7</td>
<td>5</td>
<td>7</td>
</tr>
<tr>
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<td>6</td>
<td>7</td>
<td>6</td>
<td>7</td>
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<tr>
<td>Biological</td>
<td></td>
<td>5</td>
<td>6</td>
<td>4</td>
<td>6</td>
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<tr>
<td>Total</td>
<td></td>
<td>6</td>
<td>9</td>
<td>6</td>
<td>9</td>
</tr>
</tbody>
</table>

What does the evidence base indicate in relation to the question posed?

Models, frameworks and critical zone observatories

Aitkenhead et al. (2011) described a process-based model (MOSES - Modelling Soil Ecosystem Services) that uses established physical processes and pedotransfer functions to provide information about a soil profile and its ability to deliver ecosystem services. Processes implemented within the model included thermal conduction, water movement, organic matter pool dynamics and gas/solute diffusion. The organic matter status of the profile was initialised using iterative runs of the RothC model (e.g. Farina et al., 2013) to determine changes in the size of different C pools. The model was validated against detailed time-series field measurements of CO2 concentration and emission, temperature and water content in a freely drained podzolic soil in Ireland and was effective at simulating these specific parameters, with statistically significant association between measured and modelled values. Overall, the model performed better at shallow depth, with lower levels of accuracy in deeper layers. The model simulates soil ES-related functions, including C sequestration, water buffering and biomass productivity. Additional processes and functions could be added to MOSES to improve the simulation of soil ES provision, multifunctionality and the effects of external drivers such as climate change and soil management.
Banwart et al. (2011) described how four Critical Zone Observatories (CZO’s) have been set up to represent the stages of the ‘soil life cycle’ (from incipient soil formation to highly degraded soils) and test soil process models that help assess ecosystem function. Research methods included detailed analysis and mathematical modelling of soil properties related to aggregate formation and their relation to key processes of reactive transport, nutrient transformation, and C and food web dynamics. Research focused on the central role of soil structure and soil aggregate formation and stability in soil processes and on understanding soil ES including their quantitative monetary valuation within the soil life cycle.

Initial results indicate that the development of favourable soil structure and other important biogeochemical properties and changes in soil C occur over a time scale of decades (Banwart et al., 2012). For example, the modelling predicted that set-aside land at the most degraded site could develop substantially improved soil structure with soil C accumulation over a period of several years. Other results demonstrate the rapid dynamics of soil C and how quickly it can be lost. Banwart et al. (2012) also discussed the development of frameworks that will help to communicate results into a more-policy relevant format using ES approaches.

O'Sullivan & Simota (1995) considered the use of different model types to predict the effects of soil compaction (soil stresses) on soil processes, soil function and, by inference, sustainability; and identified the major problems likely to be faced when linking production-focused models with environmental impact models. These problems included soil variability, the different spatial scales of compaction and the prediction of how compaction affects soil structure. They discussed the use of pedotransfer functions to estimate the effects of compaction on structure-dependent properties.

Mechanistic crop production models were considered to be more suitable than empirical models for predicting compaction effects. However, simpler water balance methods were more appropriate for large scale use. Mechanistic soil N models were likely to be more suitable than empirical ones for predicting compaction effects. The draught required for cultivation could be estimated with reasonable precision, but it was difficult to predict the effects of compaction on seedbed quality (related to biomass/crop production). The paper was included in a recent review of soil compaction modelling by Nawaz et al. (2013).

Wilkinson et al. (2014) describe a framework incorporating field/soil science and social science approaches to sustainable soil management to improve water quality and reduce flooding risk. They used a Catchment Systems Engineering (CSE) approach to manage a significant component of surface runoff generation by targeting hydrological flow pathways at source, such as overland flow, field drain and ditch function. The framework was based around engagement with catchment stakeholders and used field-based evidence to support decision-making and resulted in local flooding issues being largely addressed.

An example of a national applied research programme on sustainable soil management is GEESOIL (fonctions environnementales du sol – GESTion du patrimoine SOL, i.e. soil environmental functions – soil heritage management www.gessol.fr). Set up by the French Ministry of Ecology, the programme aimed to provide a scientific basis and appropriate tools for decision makers and environmental managers to improve the assessment of soil
sustainability and ‘multi-functionality’ and to reduce soil degradation risks (Bernoux et al. 2011). The programme resulted in new methods of sustainable soil management based on cause-effect relationships and the application of the DPSIR (Driving Forces-Pressure-State-Impact-Responses) approach.

**Working towards sustainable land management**
Cowie et al. (2011) discussed some technical and social issues related to achieving sustainable land management. They argue that despite complementarities between avoiding soil degradation, reducing greenhouse gas (GHG) emissions and protecting biodiversity, tradeoffs can still arise in the pursuit of these objectives. Identification of synergies, conflicts (e.g. potential for ‘pollution swapping’), trade-offs, interconnections, feedbacks and spillover effects among multiple objectives, drivers, actions, policies and time horizons are crucial to reconcile concerns and agendas. Once these issues are transparent, coordinated action can be put into place to develop strategies and policy measures to support sustainable land management.

**The role of biological indicators in ES delivery**
A’Bear et al. (2014) focused on a model for sustainability in terms of the community functioning of soil fungi and interactions between macro-, meso- and micro-invertebrates and nutrient cycling in woodlands rather than agricultural soil, but the results may inform models of sustainability for agricultural soils in terms of the continued delivery of nutrient cycling and decomposing functions.

De Deyn et al. (2014) investigated the effects of plant species diversity on C and N stocks; CO₂ exchange; and C and N leaching losses in two grassland soils of contrasting fertility. After 2 years, vegetation C and N and soil microbial biomass were greater in the more fertile soil and increased significantly with the number of plant species and functional group richness. The positive effect of plant diversity on vegetation C and N coincided with reduced loss of water and N leaching, which was especially governed by forbs, and increased rates of net ecosystem CO₂ exchange. Soil C and N pools were enhanced by the biomass and presence of the legumes *Lotus corniculatus* and *Trifolium repens*. The results highlight the importance of legumes and forbs in contributing towards ‘multifunctionality’ (primary productivity, water and nutrient cycling, and C storage) in low input agricultural systems.

5.4.1 Implications for policy and further research
A number of models such as MOSES and the CZO models can help assess the sustainability of soil management practices for a limited set of circumstances and range of ES. These models have capacity for expansion to include additional processes and functions to improve the simulation of multiple soil ES provision and the effects of external drivers such as climate change. This will be an important area for development to aid decision making and help define a sustainable soil in order to set specific goals and objectives for policy. Integrating the quantitative monetary valuation of ecosystem services will be another important step in the development of these models as decision support tools.

Corroboration of the long timescales of several years over which changes in soil quality occur confirms the need for soil monitoring with a frequency of measurement of between five and ten years.
Wilkinson et al. (2014) and Cowie et al. (2011) emphasised the importance of social science as well as soil science in the development of frameworks for sustainable soil management. This confirms the continued need for stakeholder engagement and interdisciplinary research to help address land-based challenges. Transparency of scientific information and the early identification of synergies, conflicts and trade-offs between different objectives and stakeholder groups is essential to reconcile concerns and agendas. This emphasises the need to report scientific research findings in a range of formats that are suitable for all stakeholders so that key groups can engage in discussion about soil management practices and sustainability.

There is a need to carry out detailed assessment of the biological diversity and functional diversity in arable soils to determine whether or not current management practices are sustainable in terms of their ability to deliver the key ES expected of agricultural soils.

5.5 What effect may climate change have on aspirational soil quality targets?

Volume and characteristics of the overall evidence base

The search and screening process identified 15 papers which were considered relevant to the question of what impact climate change may have on soil quality targets. Seven papers reported results from experiments which simulated the effect of climate change (i.e. warming and/or rainfall manipulation) and measured the impact on specific soil properties; six of these experiments focussed on the impact on soil biology and one study reported the impact on soil C mineralisation. In addition, there were eight review/discussion papers; two focussed on the impact of climate change on soil biology, three papers discussed potential impact on soil C (two specifically in relation to British uplands), and three more were general reviews of the potential impact of climate change on soil properties and functioning. In addition to the papers obtained from the literature search, three Defra reviews have specifically addressed the question; Defra project SP0538 (2005) ‘Impacts of climate change on soil functions’; Defra project SP1601, sub-project D (2010) ‘Review of current knowledge on the impacts of climate change on soil processes, functions and biota’; and Defra project SP1104 (2012) ‘The impact of climate change on the suitability of soils for agriculture as defined by the Agricultural Land Classification’.

Defra project SP1104 investigated the sustainable management of soils for food production and the implications of potential changes in temperature and soil water regimes for the pattern and distribution of different ALC grades. Changes to soil wetness and the future distribution of the best and most versatile land could have significant implications for how agricultural soils should be managed in England and Wales and could increase pressure for land use change. This would be the case particularly if changes in rainfall patterns resulted in reduced opportunity for field operations in autumn and spring. However, Defra project SP1316 (‘Identifying the soil protection benefits and impact on productivity provided by the access to waterlogged land requirements in cross compliance and exploring the impacts of prolonged waterlogging on soil quality and productivity’) concluded that current UK Climate Projections predict a later return to field capacity on average in most regions, and a fall in the number of field capacity days and the duration of waterlogging. The same project acknowledges that such projections do not take account of a potential increase in the frequency of extreme events due to climate change, including the possible, but highly
uncertain, potential foreshortening of return times for high energy storm events (Leckebusch et al., 2006; Villarini et al., 2011). The potential impact of climate change on aspirational soil quality targets and on sustainable soil management, therefore, remains highly uncertain.

The papers and Defra reviews identified here provide information that can be used to assess the likely impacts of climate change on soil properties and soil functioning. However, none of the literature identified discuss potential change in soil properties in relation to soil quality targets.

What does the evidence base indicate in relation to the question posed?

Soil biology

Climate change is expected to result in warmer temperatures and a change in rainfall distribution with wetter winters and drier summers, which have all been shown to have an impact on soil biology. Emmett et al. (2004) measured an increase in soil respiration in response to warming and a decrease in response to drought. Heinemeyer et al. (2004) and Kim et al. (2012) both measured a change in the composition of soil bacterial communities in response to soil warming. Lee et al. (2014) assessed the sensitivity of plant and soil arthropod communities to rainfall manipulation (i.e. simulated increase in winter rain and a simulated decrease in summer rain). Although no effects were observed in the first two years, in the third year declining plant biomass was associated with changes to soil arthropod communities. Iglesias Briones et al. (2009) measured a significant effect of soil warming on soil fauna, although the impact of warming differed between soil fauna functional groups, with significant decreases in the large oligochaete groups and Prostigmata mites and the redistribution of enchytraeids to deeper soil layers. In contrast, Holmstrup et al. (2013) investigated the impact of warming and repeated drought in long term (13 year) field experiments in Wales, the Netherlands and Denmark on soil microarthropods. They found that increased intensity and frequency of drought had no detectable effects on the microarthropod communities and suggested that these microarthropods may be only transiently impacted by repeated spring or summer drought.

Defra project SP0570 (Climate change impacts on soil biota - development of experimental methodology) explored the development of a methodology appropriate to assess the likely resistance and resilience of soil biological properties to climatically-based perturbations. Soil processes relating to C and N cycling, and the soil microbial (phospholipid fatty acid profiling) communities were measured. The perturbations were based on a single cycle of drying and rewetting, or one of flooding and draining, applied to six soils originating from Sweden, Scotland, England, France, Spain and Greece, representing a latitudinal European gradient. In general, soils were less resistant to drying than flooding. This was particularly the case for the microbial community profiling, which provided the most useful data satisfying the criteria of discrimination, sensitivity, ubiquity and interpretability. In relative terms, there was some evidence that the soils from Scotland and England showed least resistance generally. The authors concluded that the resistance and resilience assay had potential as an effective indicator of climate change impacts on soil systems, but that its further development would need to involve more extensive perturbations and high
throughput systems to accommodate the inherent temporal and spatial variation exhibited by soils.

A’Bear et al. (2014) conducted a meta-analysis review on the impacts of experimentally manipulated temperature, atmospheric carbon dioxide concentrations and soil moisture content on soil biology. The meta-analysis showed that in warmer, wetter conditions mycelial growth and mycophagous invertebrate abundance are likely to increase. De Vries and Shade (2013) argued that knowledge of what controls the stability of soil microbial communities is pivotal for predicting the impacts of climate change, and they proposed a framework for predicting soil microbial communities response to climate change based on specific functional traits present in the community.

Defra project SP1601 noted that the majority of studies to date which have investigated the impact of climate change on soil biology have considered single factors, such as elevated carbon dioxide, warming and drought. However, there is potential for interactions between these factors to have additive or antagonistic effects on soil biology and very little is known about the influence of multiple and interacting climate drivers on soil biology. Furthermore, in their review Wolters et al. (2009) observed that there is currently limited knowledge on the extent to which the biota below ground and the functions they perform are dependent on the biota above ground and vice versa.

**Soil organic matter**

Soil organic matter is susceptible to changes in land management and to changes in soil temperature and moisture. The impact of climate change on soil organic matter loss is also considered in section 4.4 (e.g. Defra project SP0571 predicted relatively small changes in soil C content in England and Wales over the period 2010 to 2080). Defra project review SP0538 noted that the impact of climate change on soil organic matter is subject to considerable debate. On the one hand it is recognised that global warming and increasing carbon dioxide levels in the atmosphere can favour increased plant growth, which in turn could provide more organic matter for the soil. However, on the other hand an increase in temperatures would be consistent with an increase in the rate of decomposition and loss of soil organic matter. Papers by Hungate et al. (2009) and Dalias et al. (2001) illustrate the complex nature and interactions between processes controlling C cycling. In a meta-analysis, Hungate et al. (2009) showed that the effect of elevated atmospheric carbon dioxide concentrations on soil C accumulation increases with the addition of N fertiliser, whilst Dalias et al. (2001) found that as well as controlling rates of C mineralisation in soils, increasing temperature can also result in the production of biomass material which is more recalcitrant; a process which could ultimately result in a restriction of the positive effect of increased temperatures on soil carbon dioxide release.

Orr et al. (2008) and House et al. (2010) discuss the impact of climate change on C storage in British uplands. Orr et al. (2008) cite evidence that direct temperature rises explain 12% of the observed increase in stream water DOC in an upland peat bog in the Pennines and repeated droughts and an enzyme latch mechanism (changes in moisture conditions affecting the inhibition of enzymes that decompose SOM) may account for the remainder. However, over longer time scales, persistent lowering of peatland water tables as a result of drought does not always result in reduced C storage. In their review, House et al. (2010)
reflect that whilst we have been able to identify and quantify the exposure of upland areas and blanket peat to climate change, we have much less knowledge about the dynamic response of their vegetation and C balance, and current model projections do not give clear results as to how climate change will affect C storage.

Soil structure and soil erosion

Climate change may also be expected to impact on soil structural condition and rates of soil erosion (section 4.4, e.g. Defra project SP0571 predicted an increase in (spatially averaged) erosion in general to the end of the century due to increased winter rainfall). However, none of the papers identified here, as relevant to this question, addressed the potential effects of climate change on soil physical properties; unlike Defra reviews SP0538 and SP1601, which did so. These projects concluded that a decline in soil organic matter, as could potentially occur as a result of climate change, might lead to a decrease in soil aggregate stability, an increase in susceptibility to compaction, lower infiltration rates, increased surface runoff and hence an increase in susceptibility to erosion. Furthermore, an increase in extreme weather events may increase the risk of soil erosion by rainfall (amount, frequency, duration and intensity) and wind (direction, strength and frequency of high intensity wind). In addition, the stability of soil could potentially be affected by the increasing intensity of wetting and drying cycles with climate change. However, it should be noted that the discussion in Defra reviews SP0538 and SP1601 was based on knowledge of how current climate affects soil properties and therefore likely impacts of climate change; the reviews did not include any studies which specifically investigated the impact of climate change on soil physical properties, and were unable to comment on the potential magnitude of any changes in soil properties.

5.5.1 Implications for policy and further research

- Defra project SP0538 reviewed the impact of climate change on soil functions and concluded that on the basis of current knowledge it is only possible to describe the likely impacts of climate change on soils in a qualitative or semi-quantitative way and highlight key changes, their direction and their implications for management.
- Knight and Harrison (2013) argued that there is a significant policy gap regarding soil and land management under climate change that needs to be closed to facilitate the sustainability of international resource use.
- Defra project SP1601 and Bardgett et al. (2013) emphasised the need for future studies on the impact of climate change on plant-soil interactions to take a holistic, ecosystem level approach in which responses of plant and soil biological communities and resulting feedbacks on nutrient and C cycling are considered together.
6. Soil management practices

6.1 Introduction
This section focuses on soil management practices to achieve a sustainable soil. Sixteen research papers addressed the question of how soils should be managed to achieve sustainability (Figure 12); nine papers explored what we mean by a sustainably managed soil and how we measure it; and three papers examined what level of intervention is needed to reach the target of sustainably managed soil. One hundred and eight papers investigated how soil management impacts on soil quality; 16 addressed how different soil types respond to management practices and whether they still have the capacity to store more C; eight explored the manipulation of soil biodiversity to improve ES delivery; and 20 addressed how climate change will affect the choice of soil management measures to achieve sustainability.

6.2 How should soil be managed to achieve sustainability?

Volume and characteristics of the overall evidence base

The search and screening process resulted in fourteen papers which directly addressed the question of “how should soil be managed to achieve sustainability”. The papers propose a number of conceptual models of global food production.

What does the evidence base indicate in relation to the question posed?

There is increasing pressure to feed a growing population estimated to reach 9 million by 2050 and increase per capita consumption of protein-rich animal produce (Alexandratos and
Bruinsma, 2012). Schulte et al. (2014) outlined that, to meet this target, increases in agricultural production are more than likely required. However, increasing production using current technology is likely to have substantial impacts on other factors e.g. drivers of global change (Gregory & Ingram, 2000).

A number of conceptual models of global food production have been developed, which address approaches for increasing food production, while providing environmental ES. Models or concepts discussed by researchers include:

- ‘Ecosystem services’ (Eigebrod et al., 2009; Everad, 2013; Hansen et al., 2005; Haygarth and Ritz, 2009; Pilgram et al., 2010; Posthumus at al., 2010; Powlson et al., 2011; Smith et al., 2013 and Smith et al., 2013b);
- ‘Sustainable intensification’ (Godfray, 2010; Gregory and Ingram, 2000; Pretty & Bharucha, 2014; Pretty, 2008);
- ‘Agro-ecology’ (Altieri, 1995; Gliessmann, 2007; IAASTD, 2009; de Schutter, 2011; The Centre for Agroecology and Food Security, 2013);
- ‘Extensification’ (Gregory and Ingram, 2000);
- ‘Climate-Smart agriculture’ (FAO, 2010);
- ‘Functional land management’ (Schulte et al., 2014);
- ‘Sustainable land management’ (Cowie et al., 2011); and
- ‘Multiple (tiered) management strategies’ (Eigenbrod, et al., 2010).

‘Sustainable intensification’ refers to producing more from the same area of land. Gregory and Ingram (2000) outlined that given that little land will be available to expand agriculture, there will be a requirement to produce more from current agricultural land, i.e. increase crop yields. Matson et al. (1997) explained that intensification results in changes in management, genotype and will require increased inputs of nutrients, water and agrochemicals. Using current available technology and practices, the main environmental impacts of sustainable intensification are an increase in total GHG, methane (CH₄) and nitrous oxide (N₂O) emissions, but reductions in emissions per unit of food produced, and potentially large offsite effects due to increase losses in nutrients and agrochemicals (Gregory and Ingram, 2000).

‘Agro-ecology’ represents the convergence of agronomy and ecology. It is the “application of ecological science to the study, design and management of sustainable agroecosystems” (Altieri, 1995). The main aim of agro-ecology is to improve the resilience and sustainability of food production systems by mimicking natural processes as far as possible (de Scutter, 2011). It involves the careful management of soil organic matter; the recycling of nutrients and energy on the farm, rather than introducing external inputs; integrating crops and livestock; and focusing on interactions and productivity across the agricultural system, rather than focusing on individual species. Agro-ecology is based on techniques developed from farmers’ knowledge and experimentation.

Lampkin et al. (2015) investigated the extent to which the sustainable intensification and agro-ecology concepts are compatible, and concluded that agro-ecology can form an integral part of achieving sustainable intensification; and that further work is required to improve understanding of the opportunities for the wider adoption of agro-ecological
systems and practices on UK farms. A reduction in the amount of food waste could provide scope for greater use of agro-ecological principles in farming. For example, 7.2 million tonnes of food and drink waste were generated in UK homes in 2010, of which 4.4 million tonnes was avoidable; representing 12% of the food and drink entering the home. (Quested et al., 2013).

Table 16. Summary of some of the key soil functions as detailed by the EU Soil Thematic Strategy; taken from Powlson et al. (2011).

<table>
<thead>
<tr>
<th></th>
<th>Description</th>
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<tbody>
<tr>
<td>1.</td>
<td>An environment for seed germination, root growth, and the functioning of roots to provide anchorage and absorb water and nutrients.</td>
</tr>
<tr>
<td>2.</td>
<td>Provision of reserves of nutrients within organic matter and mineral components, which are released into plant-available forms at different rates.</td>
</tr>
<tr>
<td>3.</td>
<td>The pathway through which water and nutrients move to roots, whether from soil reserves or from external inputs.</td>
</tr>
<tr>
<td>4.</td>
<td>The matrix in which transformations of nutrients occur through biological, chemical and physical processes, with major implications for crop uptake and losses.</td>
</tr>
<tr>
<td>5.</td>
<td>An environment for microorganisms and fauna, which may be beneficial, harmful or neutral towards crop plants. Many organisms are central to the transformations of organic matter, nutrients and pollutants with major implications for agricultural production and ecosystem processes.</td>
</tr>
<tr>
<td>6.</td>
<td>A platform for machinery, humans or animals involved in agricultural operations.</td>
</tr>
<tr>
<td>7.</td>
<td>Not moving: i.e. not being subject to erosion, mudslides or landslips and thus providing a stable surface for a range of human or natural activities.</td>
</tr>
<tr>
<td>8.</td>
<td>Absorbing water and thus retaining it for use by vegetation and transfer to rivers and streams. The opposite is surface runoff in which water moves rapidly to rivers, and ultimately to oceans, with little replenishment of soil water storage and increased risk of soil erosion and transfer of sediment to surface waters.</td>
</tr>
<tr>
<td>9.</td>
<td>Influencing water quality, positively or negatively, by regulating the transformations and movement of nutrients, pollutants and sediments to surface- or ground-waters.</td>
</tr>
<tr>
<td>10.</td>
<td>Influencing the composition of the atmosphere particularly through acting as source or sink for several GHG’s.</td>
</tr>
<tr>
<td>11.</td>
<td>Providing a habitat for soil biota which represent a vast source of biodiversity.</td>
</tr>
<tr>
<td>12.</td>
<td>Providing a basis for natural or semi-natural vegetation which, in turn, provides habitat and resources for the animal kingdom.</td>
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Other papers addressing/proposing frameworks for food production include Ostergard et al. (2009), Postschin & Young (2013), Smart et al. (2011), Sydorovych et al. (2009), Snyers (1997) and Sutherland et al. (2014).

Powlson et al. (2011) reviewed (as part of the Foresight programme on “global food and farming”) some of the key soil functions (as presented in the EU Soil Thematic Strategy) essential for agricultural soils to delivery key ecosystem services (Table 16).
Other researchers have summarised the key functions of soils by defining broad categories (Bouma et al., 2012; Haygarth and Ritz, 2009, Schulte et al., 2014 & 2011 - Table 17). Powlson et al. (2011) highlighted that managing soils for food production can be detrimental to other ecosystem services. Therefore, a priority for soils research is to identify management practices that avoid irreversible damage to the soil (section 4 soil degradation processes). Schulte et al., (2014) discuss how soils differ in their capacity to deliver each of these functions and that this is influenced by their chemical, physical, pedogenetic characteristics as well as the agroclimatic environment (Eliasson et al., 2000 and Schulte et al., 2012). Furthermore, Schulte et al. (2015) outline how the capacity of different soils to perform each of the 5 functions (Table 17) is also dependent upon land use.

The ways in which a soil can be managed to deliver particular combinations of soil functions can occur by two methods 1) direct alteration of soil properties i.e. common farm management practices, or 2) land use change (Schulte et al., 2014). Schulte et al. (2014) introduced the concept of ‘Functional Land Management’ in which the aim is to allow growth targets to be met whilst minimising the impacts on the environment. Essentially soils are managed to ensure they perform the functions to which they are suited. Schulte et al. (2014) highlighted that, this is consistent with the view expressed by other researchers (i.e. Haygarth and Ritz, 2009; Benton, 2012). Interestingly, Schulte et al. (2015) discussed how some soil functions can be offset between geographical locations whilst others cannot, ultimately this will have implications for the spatial scale at which ‘Functional Land Management’ is best applied and may vary between soil functions.

Table 17. Summary of 5 key ecosystems services as presented in Schulte et al. (2014) building on the work carried out by Bouma et al. (2012); Haygarth and Ritz (2009); and Schulte et al. (2011).

<table>
<thead>
<tr>
<th>Function</th>
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<tr>
<td>1. Production of food, fibre and (bio)fuel, which traditionally is the soil function that provides a livelihood to farmers and associated sectors in the rural environment</td>
</tr>
<tr>
<td>2. Water purification</td>
</tr>
<tr>
<td>3. C sequestration</td>
</tr>
<tr>
<td>4. Habitat for biodiversity</td>
</tr>
<tr>
<td>5. Recycling of (external) nutrients/agro-chemicals</td>
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</table>

There is some uncertainty over how soils should be managed to reduce flooding risk. The Flooding and Land Use Jigsaw (EA, 2009) aimed to identify research gaps in determining the effect of catchment scale interventions such as good soil management on flooding risk. Research projects such as FD2114 (Defra & EA, 2004) and FD2120 (Defra & EA, 2008) were unable to demonstrate that extreme flood events can be reduced to an acceptable level through changes in rural land use and management at the large catchment scale. However, there is stronger evidence that soil management can deliver flood risk management benefits at the local catchment scale (Defra, 2008). Farmland can store floodwater to reduce downstream flood risk and washland (storage) or managed realignment areas can be designed to store excess water and slow down flood peaks.
6.3 What do we mean by a sustainably managed soil and how do we measure it?  

*Volume and characteristics of the overall evidence base*

The search and screening process resulted in thirteen research papers that addressed sustainable soil management as a concept. Of the 13 papers identified, ten explicitly covered food production, ten papers addressed water and nutrient cycling and four climate change mitigation (Table 18).

<table>
<thead>
<tr>
<th>Ecosystem service</th>
<th>Food production</th>
<th>Water and nutrient cycling</th>
<th>Climate change mitigation</th>
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<tbody>
<tr>
<td></td>
<td>10</td>
<td>10</td>
<td>4</td>
</tr>
</tbody>
</table>

Table 18. Frequency that ecosystems services are addressed by papers.

Only one abstract (*Berry et al.*, 2003) specified the broad soil type that was covered in the paper. Three papers addressed measurements to assess the nutrient status of soils, 5 papers proposed methods for assessing impacts on biodiversity, 2 papers presented methods for differentiating between land uses. Finally, one paper assessed the interaction between plant species and microbial communities.

*What does the evidence base indicate in relation to the question posed?*

While section 5.2 of this report summarises in detail research on sustainable soil and methods or assessment approaches, this section briefly outlines the papers captured by the literature searches on sustainable management of soils.

Several of the papers describe methods for assessing properties or the nutrient status of soils. *Andrist-Rangel* (2010) assessed the potassium (K) reserves in northern temperate grassland soils, using a combination of: 1) X-ray powder diffraction (XRPD), 2) aqua-regia extractable K to assess how the soils ranked in term of their long-term potential to deliver K to plants. *Berry et al.* (2003), calculated N, P and K budgets for nine contrasting organic farms (stockless versus, cattle, pig and dairy with a significant proportion of arable cropping) within the UK. This approach allowed farms with nutrient surpluses to be identified.

*Kuan et al.* (2007) proposed methods to assess biological resilience and physical resilience of soils in response to stress (e.g. heat and heavy metal stresses). Biological resilience was assessed by measuring CO$_2$ evolution from the soil. Physical resilience was assessed by measuring compression and expansion indices (i.e. void ratio changes). It was found that SOC content correlated strongly with resilience after biological and physical stresses.

*Black et al.* (2003), presented the results of the Countryside Survey 2000 (CS2000), which was the first integrated study of soil biota and chemical properties at a national scale and provides evidence to select the key robust measurements of soil quality that could be used in soil monitoring schemes to determine the typical properties of a sustainably managed soil. The methodology involved collecting soil cores, which were analysed for soil microbial and invertebrate populations. In addition, information was collected on geographical location, vegetation, land use, land cover, soil type and pollutant levels.
Studies have investigated the applicability of indicators to differentiate between land management practices. Brown et al. (2000), used topsoil characteristics to differentiate between organically, conventionally managed arable and horticultural farms. It was found that farm type could be most effectively characterised by SOM, aggregate stability, infra-red absorbance and soil pH. However, the differences in soil management practices and soil properties between farm types could not be attributed to the specific management constraints within each system. Similarly, Pretty et al. (2008), investigated the use of indicators to differentiate between different management strategies, spring versus winter cropping, reduced N fertilisers, reduced pesticide application, mixed rotations and field margin management.

Several papers proposed methods to assess the biodiversity of soils. Blouin et al. (2013) reviewed the impact of earthworms on soil function and ES and soil management methods that could exploit this. Griffiths and Philippot (2013) assessed how microbes respond to soil disturbance or environmental change in order to assess their resistance (the initial response of a soil property or function to stress) and resilience (the subsequent recovery over time following removal of the stress). They concluded that soil stability results from a combination of biotic and abiotic soil characteristics and therefore could be used to provide a quantitative measure of soil health. Harris (2009) also reported that measurements of the soil microbial community could be used to indicate the status of the soil in relation to restoration targets or to assess the effectiveness of management strategies. However, Kibblewhite et al. (2008) reported that, due to the interactions between soil processes and properties, measurement of individual groups of organisms, processes or soil properties is not sufficient to indicate the status of soil quality.

6.4 What level of intervention is needed to reach the target of sustainably managed soil?

*Volume and characteristics of the overall evidence base*

The search and screening process resulted in the identification of three papers with direct relevance to this question.

*What does the evidence base indicate in relation to the question posed?*

Stoate et al. (2009) reviewed the impacts of 21st century agricultural practice in Europe. The policy and market drivers of agricultural change since 2001 and their ecological impacts are presented. The review highlights areas for further research, focusing on mitigation measures and an overview of policy implications.

Southon et al. (2013) presented the results from a nationwide field scale study investigating the impacts of N-deposition on the structure and functioning of heathland ecosystems. Overall, it was reported that N-deposition is contributing to biodiversity loss and changes in ecosystem functioning. It was concluded that, given the continued exceedance of critical N loads, further impacts on heathland diversity and ecosystem services can be expected.

In a review, Bommarco et al. (2013) discussed the requirement for ‘ecological intensification’, which was described as “the management of service providing organisms that make a quantifiable direct or indirect contribution to agricultural production”. The
review highlights areas for further research and investment, which is required to reduce existing yield gaps.

**Implications for policy and further research**

One area of research highlighted at the project workshops, and not identified from the literature search, was the influence of land tenure on soil management practices. There is much anecdotal and some research evidence to indicate that short-term land tenancies can result in poor soil management practices and a deterioration in soil quality (e.g. Fraser, 2004). Research is therefore needed to develop land management incentives such as payment for ecosystem services (PES – section 7) from government or through land tenure agreements that result in sustainable soil management practices and the maintenance of key soil functions on rented land.

**6.5 How does soil management impact on soil quality?**

**Volume and characteristics of the overall evidence base**

Overall, 159 papers that assessed the impact of soil management practices on soil quality were identified through the literature searches. Additional papers and reports were identified from the workshops. The following section summarises this literature across five broad soil function themes:

i) Storing carbon  
ii) Providing a pathway for air, water and nutrients  
iii) Supporting biota and habitat  
iv) Provision and transformation of nutrients  
v) Influencing greenhouse gas (GHG) emissions

Where possible, the land management practice investigated and the soil property or function considered is specified. Management practices can affect multiple aspects of soil quality and function. Furthermore, interactions occur between many soil qualities or functions (e.g. C storage and soil structure). Studies typically consider more than one soil quality or management practice. Consequently, they may be listed several times

*What does the evidence base indicate in relation to the question posed?*

i) **Storing carbon (SOC)**

a) Impacts of grazing and other grassland management practices

Several researchers have investigated the impact of grazing pressure on soil quality. For example, Marriott *et al.* (2010) in two long-term experiments (at one drier and one wetter grassland site) compared the impacts of extensive grazing, abandonment and continued intensive grazing on soil parameters, plant nutrient content and ecological indicator values (i.e. Ellenburg weighted indicator values). It was found that on the wetter site, abandonment led to a reduction in soil N and C compared to intensive grazing; and on the drier site extensive grazing resulted in a build-up of soil C.

Smith *et al.* (2014) compared the impact of sheep stocking intensity (a) commercial, b) low intensity, i.e. 0.9 ewes ha$^{-1}$ yr$^{-1}$ and c) no livestock) on plant C stocks in swards dominated by the grass species *Molinia caerulea* and used the RothC model to predict long-term changes
in SOC. It was concluded that under commercial sheep stocking rates SOC stocks would decline compared to no sheep and low-intensity sheep grazing.

Medina-Roldan et al. (2012) found that while grazing exclusion resulted in a slowing down in the rates of C and N cycling, there was no detectable change in C and N stocks in the topsoil. It was concluded that ‘a certain level’ of grazing is compatible with C storage in upland grassland farming.

Other studies that have considered the impact of grassland management strategies on C storage include O’Mara (2012) and Jones and Donnelly (2004), who reviewed the impact of various management strategies on C stocks. Both studies concluded that it is possible to increase grassland C-storage through changes in management. O’Mara (2012) estimated that grazing land management and pasture improvement have a global technical mitigation potential of 1.5 Gt CO₂ equivalent in 2030.

Cui and Holden (2015), evaluated the impact of grassland management on soil structural quality (assessed using the VESS method) and microbial activity related to C and N turnover. It was found that soil structural quality (i.e. sq score) was strongly negatively correlated with both soil respiration and enzyme activity, i.e. there was a decline in microbial activity associated with C and N cycling in soils with a poor structure.

b) Impacts of organic manures and/or manufactured fertiliser

A number of studies have compared the impacts of applying organic manures or inorganic fertiliser on C storage. For example, Bhogal et al. (2009) and Jones et al. (2006) investigated the impact of organic manure application and urea on C-cycling in grassland soil by measuring soil respiration; net ecosystem exchange; and nitrous oxide and methane fluxes. It was concluded that although addition of poultry manure and sewage sludge increased SOC contents, the increases in C storage did not offset the estimated additional losses of N₂O, particularly in wet years.

Markgraf et al. (2012) assessed the micro-structure properties of soils which had received long-term application of farmyard manure and found the soils had improved soil elastic properties (when wet) due to the enhanced SOC content.

On a maize site in Italy, Monaco et al. (2008) assessed the impacts of slurry, farmyard manure and plant residue additions on soil organic matter content and soil biochemical properties. Farmyard manure (FYM) applications resulted in the greatest increases in SOM content, potentially mineralisable N and soil microbial biomass.

Other studies investigating the impact of organic manure application on C-storage include: Mariscal-Sancho et al. (2011) who assessed the impact of poultry manure and thermally dried sewage sludge on soil quality. Seven years after application it was found that poultry manure was more effective than the sewage sludge at improving soil aggregation, microbial development and C and N storage.

Paterson et al. (2011) used steady state ¹³C labelling techniques to monitor the transfer of plant derived C through the food web and the associated impact on C-mineralisation rates.
De Deyn et al. (2011) investigated the impact of fertiliser application and plant seeding on carbon storage in a long term (16 years) grassland biodiversity restoration project and also assessed the impacts on soil structure, ecosystem respiration and soil enzyme activities. The study found that biodiversity restoration practices increased C and N storage and was associated with reduced ecosystem respiration, increased soil organic matter content and improved soil structure.

Foranara et al. (2013) in a long-term (19 years) experiment assessed the impact of multiple nutrient (N, P, K, Mg) supply on grassland C-storage. Nitrogen applications alone were most effective at increasing SOC storage, while applications of all the nutrients (N, P, K and Mg) were effective at increasing plant productivity. It was concluded that nutrient supply was an important driver of grassland ecosystem functioning, which can strongly affect their ability to deliver multiple ecosystem functions.

c) Impacts of cultivation and or straw incorporation
In a long-term (>20 years) study, Hazarika et al. (2009) assessed the impact of cultivation methods (no till, chisel and mouldboard ploughing) and straw management (i.e. incorporation or removal) on SOM turnover and soil quality indicators, in a silty clay loam soil in South West England. Similarly, Van Groenigen et al. (2011) carried out a nine year experiment assessing the impact of cultivation method (i.e. non-inversion or conventional tillage) and straw management (i.e. corporation or removal) on SOC storage and compared measured values to those predicted by the RothC model. Within the top 0-30 cm it was found that SOC contents were significantly increased in the reduced tillage and straw incorporated treatment, with the lowest C-contents found on the conventional tillage with straw removal treatment. It was concluded that reduced tillage helped to increase C storage by decreasing the rate at which old SOC is decomposed. Furthermore, there was no significant effect of treatments at deeper soil depths. Consistent with this, Powlson et al. (2014) discussed the limited potential of no-till agriculture to increase SOC storage. The paper reviews experimental evidence, which demonstrates that increases in soil C were largely due to a shift in the depth distribution with more C concentrated near the surface.

Two papers discussed the environmental impacts of reduced tillage in Europe. Holland (2004) reviewed the environmental impacts of conservation tillage in Europe and considered impacts on soil structure, SOC storage and soil water holding capacity, while Soane et al. (2012), reviewed the literature on no-till cultivation in Europe and discussed the impacts on crop production and the environment.

Powlson et al. (2008) reviewed the impacts of straw management on SOC storage and in a later paper (Powlson et al., 2011) discussed the impacts on soil properties using results from long-term studies. Whist it was concluded that changes in SOC content from straw addition are small, continuous straw removal would likely lead to a reduction in soil physical quality.

d) Impacts of peatland and heathland management strategies
A number of studies have assessed the impact of drainage or drain blocking on C-storage in peatland or heathland habitats, these include. For example, Grand-Clement et al. (2013) used a conceptual model to assess the effects of peatland restoration on ecosystem services, including C storage. The model was applied to Exmoor
National Park in West England and the results demonstrated the long-term benefit of restoration in reducing C losses, increasing water storage and consequently improving water quality.

Ramchunder et al. (2009) reviewed the impact of drainage, drain-blocking and rotational heather burning in UK peatlands and assessed the environmental impacts including C storage. The impacts of rotational burning on C-storage was also assessed by Clutterbuck and Yallop (2010), who reported that in four out of six catchments studied, humic coloured DOC increased by (53-92%) and that moorland burning on blanket peat explained >80% of the increased losses in humic coloured DOC.

Marrs et al. (2007) investigated the impact of conservation management implemented to restore a Calluna vulgaris dominated moorland invaded by bracken. While conservation effects to improve biodiversity were successful, there was a large loss of C and N from the system.

Quin et al. (2015) assessed the impact of above ground plant species on SOC storage by comparing a Calluna vulgaris dominated upland heathland with a species poor graminoid sward, within three upland heaths in Scotland. It was found that the Calluna-dominated heaths sequestered 3.45± 0.96 t C ha⁻¹ yr⁻¹, which was double the amount sequestered by the grass-dominated heathland at 1.61± 0.57 t C ha⁻¹ yr⁻¹.

e) Impacts of organic versus conventional farming practices

Hathaway-Jenkins et al. (2011) collected soil samples from 16 pairs of organic and conventional farms in England, both arable and grassland. For similar soil types, the study compared a range of chemical and physical soil properties, including soil organic matter content. The results found that there was no overall effect of organic farming practices on soil conditions.

Shannon et al. (2002), compared the impacts of organic and conventional farming on the soil microbiology at two farms located in England. They reported that there were only small differences in the microbiology of differently managed soils and that parameters such as total SOC and microbial biomass C showed no consistent differences between management practices. Some of the differences reported included a greater abundance of total and active fungi within the organically managed compared to the conventionally managed soil.

In a meta-analysis of 74 pairwise comparisons of organic versus conventional farming systems, Gattinger et al. (2012) reported significantly higher values for SOC concentrations (0.18±0.06%); SOC stocks (3.50±1.08 Mg C ha⁻¹); and C sequestration (0.45± 0.21) on the organic farms. However, the main factor accounting for these differences was in the greater external C inputs and crop rotations between the organic and conventional farms i.e. management practices that can be carried out on both conventional and organic farms.

f) Impacts of field margin or buffer strip management

Fallon et al. (2004) presented a preliminary analysis of the C sequestration potential for different field margin management practices, by assessing the impacts of field margin width and management (e.g. grass strips, hedgerows and tree strips) on SOC-storage and possible
impacts on trace gas fluxes (e.g. N\textsubscript{2}O). The study concluded that management of field margins presents a significant opportunity to increase C sequestration. However further evidence on the nature and area of field margins within the UK was required to permit a more detailed analysis.

Stutter and Richards (2012) assessed the impact of vegetative riparian buffer strips on nutrient retention. The study collected soil samples from 19 buffer strips and adjacent fields, which were analysed to assess the potential for the release of particulate P and dissolved P, N and C. The study concluded that soil conditions within the buffer strip could potentially enhance the rate of nutrient leaching by increasing the solubility of C, N and P.

g) Impacts of bioenergy crops
Two studies assessed the impact of bioenergy crops on SOC storage:

Hiller et al. (2009) carried out a life cycle assessment of four different bioenergy crops: oilseed rape, Miscanthus, short-rotation coppice (SRC), willow and forest residues. The study carried out life cycle analysis (LCA) up to the ‘farm gate’, focusing on changes in SOC; impacts on energy use, emissions of GHG, acidification and eutrophication; and offsetting GHG emissions associated with the previous land-use. In summary, the study found that, Miscanthus and SRC were most likely to have a beneficial impact in reducing GHG emissions.

Brandao et al. (2011) modelled the GHG mitigation potential of different bioenergy crops: Miscanthus, short rotation coppice, poplar, winter wheat and oilseed rape. In terms of GHG mitigation potential the study demonstrated that Miscanthus was the best option since it sequesters C at a higher rate than the other crops. However, the study also highlighted that it is important to consider the overall environmental impact of the crops studied. For example, Miscanthus performed worst in terms of acidification and eutrophication.

h) Impacts of other management strategies or general review papers
Defra project SP1113 (Capturing cropland and grassland management impacts on soil carbon in the UK Land Use, Land Use Change and Forestry (LULUCF) inventory) aimed to identify the extent to which arable and grassland management practices could be incorporated into the UK’s LULUCF inventory, in terms of their impact on GHG emissions and SOC storage. The key management practices identified for arable land were crop type, crop residue returns, manure and fertiliser inputs and tillage regime; and for grassland were grassland type, residue returns, manure and fertiliser inputs, rotation pattern and erosion.

The review concluded that reduced tillage was not a reliable management option for increasing SOC stocks in UK soils. However, increasing crop residue returns, crop rotation and increasing inputs of manure and fertiliser could increase SOC stocks through addition of organic material and increased crop yield, although the SOC stock increases resulting from manure and fertiliser inputs could be outweighed by increases in nitrous oxide emissions and nitrate leaching losses.

Overall the impact of arable land management on SOC was likely to be very small compared to other activities in the LULUCF inventory such as land use change. The most effective mitigation option was arable reversion to grassland or forestry. However given the need for
food production there is limited scope for such change. A lack of field data on the effect of grassland management and improvement on SOC stocks was identified as a knowledge gap.

Other studies investigating the impact of land use or changes in land use on SOC storage include Hirsch et al. (2009) in which the impact of fallow land on soil microbes and SOM storage was assessed and Bending et al. (2000 & 2004) who assessed the impact of land use and land use change on SOM storage and quality. A number of other studies have reviewed or considered the impact of a range of land management practices on SOC storage or soil quality include. For example, Eigenbrod et al. (2010) assessed whether multiple conservation management strategies increased or decreased ecosystem services, e.g. C storage; Fallon et al. (2002) used the dynamic SOM models RothC and CENTURY to assess the potential of different arable land management techniques to sequester C; Freibauer et al. (2004) reviewed the potential for agricultural soils in Europe to sequester C by considering both the technical and economic implications of any management practices implemented to enhance C-storage; Glenk & Colombo (2011) reviewed issues related to the development of soil C sequestration policies into agri-environmental schemes; Gosling et al. (2013) carried out a meta-analysis to assess the effect of land management factors and environmental variables on particulate organic matter (POM), light fraction organic matter (LFOM) and total OM; and Gregory et al. (2009) in a laboratory-based study tested the hypothesis that the resilience of soil to stresses is in part controlled by the SOM content, by testing a range of arable and grassland soils from 15 sites across England.

ii) Providing a pathway for air, water and nutrients (soil structure)

a) Impacts of grazing and other grassland management practices

Cui et al. (2014) compared soil structural quality over a range of land management intensities. Typical ranges of grassland management intensity did not result in significant soil structural damage. However, there was some indication that greater intensification did lead to a degree of soil structural degradation. The impact of stocking intensity was also investigated by Sansom (1999), who discussed how livestock in the uplands have contributed to soil erosion, riverbank erosion and increased runoff.

Defra project BD5001, investigated soil structural degradation under grassland and measures to ameliorate its impact on biodiversity and other soil functions. The study assessed soil structural condition in 300 grassland fields (sward age > 5 years) in England and Wales using measurements of soil bulk density (BD), sheer strength, penetrometer resistance, Landcare Visual Soil Assessments (VSA) and Peerlkamp soil structure assessments. There was a small, but significant difference \( (P<0.01) \) in BD values (0-10 cm depth) between the ‘mainly cut’ (mean = 1.02 g cm\(^{-3}\)) and ‘mainly grazed’ (mean = 0.95 g cm\(^{-3}\)) fields. Also, 0-10 cm BD \( (P<0.001) \) was highest in improved grasslands \( (P<0.001; \text{mean } = 1.02 \text{ g cm}^{-3}) \) and lowest in semi-improved and species rich grasslands \( (\text{mean } = 0.88 \text{ g cm}^{-3}) \).

However, stocking rate at the farm and field scale were not good predictors of (0-10 cm) soil BD (Newell Price et al., 2013). This may be because ‘livestock unit per hectare’ is a crude measure of actual compaction risk. The latter relates more to the timing of loading (machinery or livestock) than the actual forces applied. However, the data also indicates that higher organic matter content and older swards can impart a certain level of resilience to compaction in the surface soil layer.
Field experiments carried out as part of Defra project BD5001 demonstrated that on medium soils in moderate condition (Mueller et al., 2010) mechanical loosening to 30-35 cm depth resulted in saturated infiltration rates that were 4-to 10-fold greater than on the unloosened control \((P<0.001)\) with effects persisting for 2-3 years post-loosening. This increase in water infiltration rates compares with a 51% reduction in surface runoff using a hybrid of perennial ryegrass and meadow fescue (‘Festulolium cv. Prior’) at the laboratory scale (Macleod et al., 2013) and a c. 60-fold increase in median water infiltration rates in the fifth year post tree planting compared with grazed pasture at the plot scale (Carroll et al., 2004; Marshall et al., 2014), and could have significant implications for the management of grassland soils in reducing surface runoff and flooding risk, particularly in catchments dominated by dairy and beef/sheep farms. However, BD5001 also found at a single site that mechanical loosening also had a significant impact on earthworm numbers and biomass with the effect from shallower loosening (c. 20 cm depth) persisting into a second year. Although the effect was measured at a single site, this could have significant implications for the regular use of mechanical loosening and work at other sites and soil types is needed to confirm the effect. Interestingly the introduction of deep-rooting herbs and legumes had no effect on earthworm numbers or biomass, but the cultivation effect associated with establishing the seed mix did result in reduced water infiltration rates in the first year and no effect thereafter.

Gregory et al. (2010) assessed the impact of different forage grasses and their rooting depth on soil structural quality, water repellence and water release characteristics. However, overall no differences could be detected between grass species. By contrast, Macleod et al. (2013) did measure increases in water infiltration rate and decreases in surface runoff in pot experiments using Festulolium (perennial ryegrass and fescue) hybrid grass species. The ongoing LINK-BBSRC 5-year funded project SUREROOT follows on from this work, using hydrologically isolated plots and work on 8 commercial farms to determine if soil structure can be modified by the root architecture of hybrid grassland species to reduce flooding and drought while maintaining productivity.

b) Impacts of organic manures and manufactured fertiliser

Griffiths et al. (2010) investigated the effects of cattle slurry and green-waste compost application for one year on soil structural quality which was assessed using the VSA method (Ball et al., 2007). Overall this study reported no improvement in soil structural quality following the application of organic materials. Mariscal-Sancho et al. (2011), also investigated the impact of applying organic manures on soil structural quality (see section 6.5 i). Markgraf et al. (2012) compared the impact of organic manure and manufactured fertiliser application on soil structural quality (section 6.5i), while in a grassland biodiversity restoration project, De Deyn et al. (2011) assessed the impact of manufactured fertiliser application on soil structural quality (section 6.5 i).

i) Impacts of cultivation and or straw incorporation

Hazarika et al. (2009) evaluated the impact of cultivation method and straw management on soil structural quality in a long-term (i.e. more than 20 years) field study. They found that no-till and chisel ploughing increased soil C in the surface horizons and improved soil
structural stability in surface layers compared with mouldboard ploughing. However, mouldboard ploughing distributed C more uniformly throughout the soil profile.

Studies investigating the impact of cultivation method on soil structure include: Ball et al. (2007) who investigated the impacts of both grazing duration treatment and ploughing timing (either mid-winter or early spring ploughing) on soil structure, SOM content and plant root growth in first spring barley crop after a grass-clover ley. In summary it was found that macroporosity was the best indicator of soil structural quality because it was a sensitive indicator of compaction and root growth.

Da Silva et al. (2014) compared soil structural quality between the row and interrow in no-tillage systems. Overall, soil conditions were less favourable in the interrow and the action of coulters when drilling improved conditions within the row.

Dexter and Bird (2001) developed a method to predict the range of soil water contents at which tillage can be carried out; Komas et al. (2001), assessed the impact of ploughing depth on soil displacement and found that tillage at a plough depth of 18 cm resulted in 60% lower soil movement compared with tillage at 25 cm plough depth. Askari et al. (2013) used the VESS method to assess the impact of conventional and minimum tillage, and crop rotation versus mono-cropping on soil structural quality. Soil quality scores (Sq) were slightly lower under minimum tillage (Sq = 1.9) compared to conventionally tilled (Sq = 2.29) soils, indicating that minimum tillage resulted in a slight improvement in soil structure.

Holland (2004) and Soane et al. (2012) reviewed evidence from across Europe on the environmental impacts of conservation tillage, while Morris et al. (2010) reviewed the agronomic impact of non-inversion tillage in the UK.

Powlson et al. (2014) reviewed the impacts of no-tillage on SOC storage (section 6.5 i); and Powlson et al. (2008 & 2011) considered the impact of straw incorporation on soil structural quality (section 6.5 i).

c) Impacts of peatland or heathland management
A number of studies have assessed the impact of drainage or drain blocking on soil structure (either on water storage capacity or impacts on water quality) in peatland or heathland habitats. These include: Carroll et al. (2011), Grand-Clement et al. (2013) and Ramchunder et al. (2009 & 2012) (Section 6.5 iv). The impacts of combinations of managed burning and grazing intensities on water table depth was assessed by Clay et al. (2009), while Ramchunder et al. (2009) also assessed the impacts of rotational burning on soil structure.

d) Impacts of field margin or buffer strip management
Carroll et al. (2004) assessed the impact of tree shelterbelts on flood risk; while, Owens et al. (2007) investigated the impact of contrasting grass buffer strips on flood risk and water quality. Overall the results demonstrated that grass buffers can improve soil structure, resulting in dramatic increases in water infiltration rate at the plot scale (up to c. 60-fold) in the fifth year post tree planting compared with grazed pasture.
e) Cover crops
Cover crops can improve soil structure in tillage farming systems. A number of Defra funded studies and research papers have investigated the environmental benefits of using cover crops mainly focusing on impacts on diffuse water pollution (e.g. NT0402; SP0404; WQ0140; Davies et al., 1996; Shepherd and Lord, 1996). However, cover crops can also provide biodiversity and soil protection benefits through improving soil structural quality or maintaining SOM; all of these issues including practicalities for farmers, were discussed at a recent AAB conference (Aspects of Applied Biology, 129, 2015). Stobart et al. (2015) as part of the Kellogg’s Origins programme assessed soil structural quality using the VESS method. Soil structure was assessed in autumn and again in spring. Where cover crops were used it was found that soil structure had improved. Abdolahi et al. (2014) also reported improved soil structural quality following 5-years of cover crop use.

As part of the New Farming Systems (NFS) research project a range of cover cropping approaches are being evaluated including the use of long term clover bi-crops and a brassica and legume mix used ahead of spring sown crops in the rotation. The project has demonstrated improved soil characteristics and yield responses with the use of cover crops in some years (Stobart and Morris, 2014; Stobart and Morris, 2013).

f) Impacts of other management strategies or general review papers
A number of studies have investigated the impact of machinery on soil structure:

Douglas et al. (1998) assessed the impact of converting from 1) conventionally managed traffic to a reduced traffic system and 2) a reduced traffic to zero traffic system. In both cases, switching the traffic management system led to improvements in soil structure, measured as increases in the volume and frequency of macropores and reduction in shear strength.

Deasy et al. (2010) as part of the Defra-funded project PE0206 – “Field testing of mitigation options” - assessed the impacts of various mitigation strategies on soil erosion and water quality. The project found that tramlines can be a major source of sediment losses and that tramline management and the associated improved soil structure can help reduce surface runoff volumes and sediment losses by 72-99%.

Ball et al. (2005) reviewed the impact of crop rotation on soil structural development and preservation, and considered how soil structure might have other beneficial effects upon nutrient cycling and disease suppression.

Glendell et al. (2014) addressed the impact of land use on soil structure, by assessing soil structure and water quality; and Glendell et al. (2014) reported that arable and intensive grasslands were ‘high-impact’ land uses, associated with negative impacts on soil structure and increases in diffuse water pollution. Whereas moorland was defined as ‘low impact’ land-use and was associated with improved water quality.

Hathaway-Jenkins et al. (2011) compared the impacts of organic and conventional farming on soil structural quality. Studies reviewing or considering the impact of a range of land-
management practices on soil structure include: Evans (2005); Hess et al. (2010), Holman et al. (2011); Locks Guimaraes et al. (2013) and Spurgeon et al. (2013).

iii) Supporting biota and habitat
a) Impacts of grassland grazing
A number of studies have investigated the impact of grazing on the biodiversity of above ground vegetation, these include: Mariott et al. (2010), Medina-Roldan et al. (2012) (section 6.5 i).

Sansom (1999) discussed how increases in sheep stocking density had resulted in a decrease in upland biodiversity. In a special issue of the Journal of Ecology (volume, 38, 2001), the impact of grassland grazing on biodiversity was addressed, including the impacts on soil invertebrates.

De Deyn et al. (2011) assessed the impact of a long-term biodiversity restoration project on the rate of soil C and N storage (section 6.5 i). Other studies investigating the impact of grassland management strategies on biodiversity include Critchley et al. (2004) who in a lowland semi-natural grassland found that restoration of botanical biodiversity coincided with low fertiliser input or changes in grazing intensity.

Marriott et al. (2004) reviewed the results from long-term experiments investigating the impacts of extensification (i.e. reducing fertiliser inputs and stocking density) of grassland management on biodiversity and productivity in upland regions. Overall, extensification resulted in benefits for biodiversity, but a reduction in productivity.

Cui and Holden (2015) investigated the relationship between soil structural quality and microbial activity (section 6.5 i).

Hartley et al. (2005) investigated the effects of a 6-year long experiment of N, P and K additions and protection from grazing on an upland heather moorland. Overall it was found that the impact of N addition on the cover of Calluna and competing grass species was largely driven by the level of grazing.

b) Impacts of organic manures and manufactured fertiliser
Stark et al. (2007) compared the impacts of short-term management practices (i.e. green manure and manufactured fertiliser application) on microbial diversity. Green manure application improved soil biology by increasing microbial biomass and activity. However, microbial community structure was found to be influenced by inherent soil and environmental factors rather than short-term nutrient inputs.

Studies investigating the impact of organic manure application on microbial diversity include Harris et al. (2011) who hypothesised that the immediate responses of microbial communities to organic manure application is governed by the long-term site history of organic manure application. Mariscal-Sancho et al. (2011) reported that poultry manure application was particularly effective at increasing microbial development. Paterson et al. (2011) found that applications of municipal green compost rather than cattle slurry had an impact on soil microbial community structure and increased nematode abundance.
Two studies (Cole et al. (2005) and Fountain et al. (2008) have investigated the impact of nutrient inputs on soil food webs reporting that ‘bottom-up effects’ (i.e. increased food resources for soil fauna) drive changes in mesofauna community structure. Cole et al. (2005) found that additions of calcium and N increased the density of microarthropods (mites and Collembola). While, Fountain et al. (2008) reported that N and lime applications increased the number of juvenile Collembola and changed the community structure of spiders which was most likely a consequence of increased food resources.

Griffiths et al. (2010) assessed the impacts of organic material, application on soil quality, by measuring abundance and biodiversity of soil biota and a range of biological, chemical and physical characteristics. Thiele-Bruhn et al. (2012) reviewed the impact of soil management practices on soil biodiversity and highlighted that manufactured fertiliser and agrochemicals have a strong influence on both the biodiversity and functioning of soil biota. Smith et al. (2008c) investigated the impacts of grassland restoration management strategies, including applications of fertiliser and farmyard manure on the biodiversity of above ground vegetation.

c) Impacts of cultivation and or straw incorporation
Holland (2004) reviewed the environmental impacts of conservation tillage within Europe (section 6.5 i and ii), and considered the impacts on soil biota (macro, meso and microfauna). Overall, studies demonstrated that conservation tillage (CT) can improve the abundance of soil biota, which in turn can improve nutrient cycling and may also help control crop pests and diseases. Brennan et al. (2006) reported that conservation tillage increased the abundance of most Collembola species, but had little effect on species richness. It was also found that straw incorporation did not increase the diversity of Collembola and reduced their overall abundance. Powlson et al. (2011) considered the impacts of straw incorporation/removal on a range of soil properties including the soil microbial community.

d) Impacts of organic versus conventional farming practices
Two studies (Orr et al., 2011 and Shannon et al., 2002) have compared the impacts of organic and conventional farming on the soil microbial community. Orr et al. (2011) discussed how management regimes (rotation, crop protection and fertility management) associated with conventional versus organic farming affected the total bacterial community and the free-living N-fixing diazotroph community. By contrast, Shannon et al. (2002) reported that there were few differences in total C and microbial biomass C between organic and conventional systems. However, the results also indicated that organic management can lead to a greater number of viable but non-culturable soil microorganisms.

e) Impacts of peatland or heathland management strategies
Grand-Clement et al. (2013), developed a conceptual model to evaluate the impacts of peatland restoration on ecosystem services, including biodiversity (section 6.5 i), while Marrs et al. (2007) assessed the impact of conservation management strategies on biodiversity (section 6.5 i). Anderson et al. (2013) reviewed the factors that determine the structure of microbial communities in natural and disturbed peatlands and the response to anthropogenic and natural disturbances.
f) Impacts of field margin or buffer strip management

Stutter and Richards (2012) investigated the differences in soil quality including microbial diversity, between riparian buffer strips and field soils. Smith et al. (2008a & 2008b) reported greater soil macrofauna biodiversity within grassy strip margins compared to within the crop. Cole et al. (2008) assessed the impacts of contrasting buffer strips on carabid biodiversity; overall, buffer strips were found to increase carabid biodiversity.

g) Impacts of other management strategies or general review papers

Researchers have investigated the impact of other management practices on soil biodiversity. The effect of land use, crop type or change in cropping has been investigated by Hirsch et al. (2009), Bending et al. (2000 & 2004), Dauber et al. (2010), Blackwell and Pilgrim (2011) and Hilton et al. (2013).

Bullock et al. (2011) and Douglas et al. (1998) investigated the impact of reduced ground pressure on soil macrofauna. Macdonald et al. (2004) assessed the impact of defoliation and soil amendments on soil microbes. Jeffries et al. (2003) and Sayer et al. (2013) investigated the impact of soil management strategies on bacteria and AMF communities. Barrios et al. (2007) and Jackson et al. (2007) addressed the issue of functional diversity. Spurgeon et al. (2013) reviewed the impact of land use and land management on earthworms. Gregory et al. (2009), as part of a study assessing the impact of long-term management on the resistance and resilience of soils, studied effects on soil mesofauna. Finally, Eigenbrod et al. (2010) reviewed the impacts of conservation strategies on soil biodiversity.

iv) Provision and transformation of nutrients

a) Impacts of grazing and other grassland management practices

Studies which have investigated the impact of stocking density or grazing pressure on soil nutrient content and turnover include: Marriott et al. (2010) and Medina-Roldan et al. (2012) (section 6.5 i). Other studies investigating the impact of grassland management on the availability and release of nutrients from agricultural soils include Luescher et al. (2014) who reviewed the use of legumes within grassland-livestock systems and the potential benefits (e.g. reduced need for manufactured fertiliser N-inputs); evidence suggesting that a mixed sward consisting of 30-50% of legumes is most effective. Perring et al. (2009) investigated methods for depleting soil P contents in order to restore plant diversity, these included applying manufactured N fertiliser followed by cutting to remove the plant biomass. Edwards and Withers (1998), reported that large farm P-surpluses (>20 kg ha⁻¹) are typically associated with intensive-livestock production. It was found that increases in surplus-P, did not always correspond to increases in total and extractable soil P, and that losses of P from agricultural soils is controlled by factors other than the size of the annual P-surplus.

b) Impacts of organic manures and/or manufactured fertiliser

Understanding the contribution of organic manure applications to crop nutrient requirements is fundamental to reducing pollution losses, without this information, farmers risk over applying nutrients and increasing nitrate, phosphorus losses to water and ammonia and nitrous oxide emissions to air (Nicholson et al., 2013).
A number of experiments have improved our understanding of the crop nutrient supply from organic manures, and have investigated methods to maximise nutrient availability whilst minimising losses to the environment. Slurry-NR - slurry management strategies to minimise N losses (Defra project ES0114), evaluated the effects of slurry application rate and method (broadcasting and bandsprea/shallow injection) on N emissions (ammonia, nitrous oxide and nitrate) to the air and water environments. Opti-N (Defra project NT2001) was carried out on demonstration farms to promote the environmental and economic benefits of improved manure management practices on commercial farms.

Sagoo et al., (2013) as part of a HGCA industry funded project (report no. 522) quantified the sulphur supply from spring and autumn applications of organic materials to winter wheat crops. It was found that for spring applied organic materials, ‘extractable’ S (i.e. readily available SO₃) was a good indicator of crop available S, ranging from 15% of total SO₃ for cattle FYM to c.60% of total SO₃ for broiler litter. Lower S use efficiencies were recorded for autumn applied organic materials because readily available S was lost due to overwinter leaching.

The decision support tool MANNER-NPK (Nicholson et al., 2013). was developed (with Defra funding) using evidence from experiments to predict crop nutrient (N, P, K and Mg) supply by taking into account factors such as manure nutrient content, mineralisation of organic N, application method, application timing and pathways for nutrient losses including, NH₃ volatilisation and NO₃ leaching.

DC-Agri is a four year project (funded by WRAP, Zero Waste Scotland, Defra and the governments of Scotland and Wales) which investigated the use of anaerobic digestate and compost in agriculture. To date, the project has produced a number of bulletins (http://www.wrap.org.uk/content/digestate-compost-agriculture) which summarise the key findings of the project and give good practice guidance for the use of digestate and compost.

As part of DC-Agri, field experiments were carried out in which ammonia, nitrous oxide and nitrate leaching losses were measured following the application of digestate and compost. It was recommended that precision application methods (i.e. bandsprea/shallow injection) are used when applying digestate to minimise ammonia loss and maximise the amount of N taken up by the crop. The DC-Agri experimental sites have demonstrated that applications of organic materials (compost, digestate and livestock manures) increased crop yields by 0.2-1.56 tonnes per hectare and resulted in N fertiliser savings, together worth £60-380 ha⁻¹. The project also demonstrated the benefits of medium and long-term use of compost and other organic materials. For example, applications of green compost over nine years at two experimental sites increased SOM levels by over 20% and reduced soil shear strength by 5%, indicating that compost application could make cultivations easier, potentially delivering fuel cost savings.

Dungait et al. (2012), reviewed nutrient management within UK agriculture. The review highlighted strategies that require further development to help improve nutrient efficiency, these include more efficient use of manufactured fertilisers, increased recovery and
recycling of waste nutrients and improved use of inorganic and organic nutrient reserves in the soil.

Del Prado et al. (2008) used a modelling framework system (Sustainable and Integrated Management Systems for dairy production, SIMS (DAIRY)) to assess the potential of management strategies to improve nutrient efficiency. Optimising manufactured mineral-N fertiliser application rate and timing was the only management strategy which reduced the environmental impact whilst being economically viable. Petersen et al. (2007) and Whitmore et al. (2012), reviewed the importance of nutrient management to reduce pollution, from both organic manures and manufactured fertilisers, while Sylvester-Bradley and Withers (2011) and Withers et al. (2014) discuss the potential areas for crop nutritional innovation which could help improve nutrient use efficiency. For example, while not directly related to soil quality, research has been carried out on the modification of nitrogen fertiliser through the use of inhibitors (i.e. urease or nitrification inhibitors) with the aim of reducing N-losses and increasing nitrogen use efficiency. A number of researchers have carried out research to evaluate the efficacy of urea or urea ammonium nitrate (UAN) treated with the urease inhibitor N-(n-butyl) thiophosphoric triamide (nBTPT) e.g. Chambers and Dampney (2009) as part of NT26; and Watson et al. (2009) discussed the mode of action, efficacy and economics of urease and nitrification inhibitors. Misselbrook et al. (2014) summarise the results from a UK research programme consisting of 14 experiments (Defra project AC0213) in which, the effect of nitrification inhibitors on ammonia, nitrous oxide, nitrate leaching losses and crop yields was investigated, following applications of manufactured fertiliser or grazing returns (i.e. urine & dung).

c) Impacts of other management strategies or general review papers
A number of other studies have investigated the effects of soil management practices on the provision and cycling of nutrients in agricultural soils. For example, Favaretto et al. (2012) assessed the impact of gypsum application on P and N losses; Hathaway-Jenkins et al. (2011) compared the amounts of plant available nutrients in organic and conventional farming systems; and Andrist-Rangel et al. (2007) investigated K dynamics in organic and conventional farming systems.

Most modern crops have unfulfilled yield potentials with no yield increases at the regional or national scales observed for 2 decades. To help close the yield gap, research and knowledge exchange programmes are being carried out to investigate and share soil and crop management practices and technologies for improved crop production. The interaction between soil resources and root properties and how this determines water and nutrient uptake are being assessed as part of the Rothamsted 20:20 wheat programme (http://www.rothamsted.ac.uk/our-science/2020-wheat), which aims to increase wheat yields to 20 t ha\(^{-1}\) within 20 yields.

Freestone (2013), through the Nuffield farming Scholarship (http://nuffieldinternational.org/rep_pdf/1413190469Jake-Freestone-2013-report.pdf) investigated technologies and management practices implemented by wheat growers around the world to increase yields. To break through the yield plateau in the UK it was recommended that improvements in soil health and structure are needed to ensure resilience to changing weather conditions and allow crop varieties to reach their yield
potential. Freestone (2013) propose that this can be delivered by focusing efforts on 4 areas: zero tillage, diverse crop rotations, cover-cropping and livestock integration.

Ongoing studies investigating developments in technology include:

- The ‘Precision Farming in Horticulture’ project (PF-Hort - The application of precision farming technologies to drive sustainable intensification in horticulture cropping systems; AHDB Horticulture CP107a) will evaluate the current and future potential of precision farming techniques to improve soil and nutrient management in horticulture. It will include a soil structure survey to assess soil structural condition under horticulture cropping systems and establish a baseline of current soil management practices; and will review available precision farming techniques and their potential application to horticulture. This will be followed by demonstration and evaluation of techniques with the greatest potential in field experiments on six commercial farms.

- SoilSense (Innovate UK project 101624 - ‘Real time measurement of primary soil nutrients for efficient crop production and management) aims to deliver an in-soil sensor for continuous monitoring of available moisture, pH and nutrients at multiple soil depths. The sensor will allow ‘smart’ dynamic control of fertiliser application in order to optimise nutrient inputs.

- Agronomics (Innovate UK project 101627 - ‘Spatial experimentation to transform research in agronomy worldwide’) has the twin aims of improving precision and extending the scale of agronomic testing and experimentation, so that farmers, advisors, suppliers, researchers and regulators will all be able to detect and aggregate small, as well as large, effects of treatments on crop performance and their interactions with soil type.

- Tru-Nject (Innovate UK project 101822 - ‘Proximal soil sensing based variable rate application of subsurface fertiliser injection in vegetable/combinable crops’) aims to precision manage variation in soils in order to stabilise crop yields and reduce fertiliser inputs. The project will use engineering, sensor, satellite image data and fertiliser placement technologies to apply N-fertiliser below ground to improve resource efficiency and reduce the N2O-emissions released from N-fertilisers.

- Soil-for-life Beta (SfL) (Innovate UK project 101822 - ‘Optimising big data to drive sustainable intensification’) is a 3-year project, within the interdisciplinary field of ‘agri-informatics’. The aim is to carry out an in-depth analysis of big data sets to provide scientific evidence to support the sustainable intensification and maintenance of soil health at a field, farm and enterprise scale. The project will develop a soil information system to map, assess and monitor soil resources to better understand the relationships between the cropping environment, soil health and sustainable, profitable agriculture. The project output will allow SfL to be rolled out as a fully functional product.
v) Influencing greenhouse gas (GHG) emissions

a) Impacts of grazing and other grassland management practices
Defra project BD5001 reported that mechanical loosening of ‘high’ bulk density medium textured grassland soils had no effect on initial or cumulative nitrous oxide emissions following the application of manufactured N fertiliser at a an application rate of 40 kg N ha\(^{-1}\). By contrast, Hargreaves et al. (2013) found that mechanical compaction (using two passes with a 10 tonne tractor) and trampling by cattle increased cumulative nitrous oxide emissions by 29% compared with the control (1,215 N\(_2\)O g N ha\(^{-1}\) cf. 944 N\(_2\)O g N ha\(^{-1}\)). However, the differences in findings between the two studies are probably due to one investigating the effects of mechanical loosening on nitrous oxide emissions in fields with ‘typical’ levels of soil structural degradation (i.e. alleviating compaction) and the other investigating the effect of tractor compaction and livestock trampling treatments on nitrous oxide emissions (i.e. causing compaction).

Luescher et al. (2014) reviewed the impacts of including legumes within grasslands, which could have potential benefits in reducing greenhouse gas emissions by reducing the dependence on fossil fuels. O’Mara (2012), discussed the role of grasslands in providing food security and mitigating climate change. Horrocks et al. (2015) measured N\(_2\)O emissions from recently created grassland (< 10 years old) converted from intensive arable land, generally N\(_2\)O emissions were less than 50 g N ha\(^{-1}\) day\(^{-1}\) across all management types.

b) Impacts of organic manures and / or inorganic fertiliser
Gibbons et al. (2014) assessed the impact of liming and nutrient management on GHG emissions. It was estimated that liming soils to pH 6.0 would reduce both N-leaching and N\(_2\)O emissions. However, liming also increases CO\(_2\) emissions resulting in a net negative impact upon climate change.

Jones et al. (2006) assessed the impacts of organic manure and manufactured fertiliser applications to grassland on CH\(_4\) emissions; it was concluded that the benefits of increased C sequestration on poultry manure and sewage pellet treatments were outweighed by additional losses of N\(_2\)O, particularly in wet years.

Petersen et al. (2013) reviewed global manure management practices for GHG gas mitigation. While the review focused on the impacts of storage and treatment technologies (e.g. slurry separation), the impacts of land spreading and the use of NIs were also discussed.

As discussed above, numerous studies have investigated the use of inhibitors to reduce N-losses. Misselbrook et al. (2014) found that, while the nitrification inhibitor Dicyandiamide (DCD) could be very effective at reducing N\(_2\)O emissions, it had little effect on NH\(_3\) volatilisation, NO\(_3\) leaching losses, crop yield or crop N offtake. Based on the reduction efficiencies results from 14 field experiments carried out in England, it was concluded that, “an approximate 20% reduction in N\(_2\)O emissions from UK agriculture is technically feasible with little risk of increasing NH\(_3\) emissions”.

99
c) Impacts of cultivation and or straw incorporation

Powlson et al. (2012) (as part of Defra project SP0561) reviewed the impacts of reduced or zero tillage on C storage and GHG emissions. In summary, it was concluded that, reduced tillage in the UK or North-West Europe is unlikely to significantly contribute to climate change mitigation, due to: 1) a small accumulation of SOC due to agronomic climatic conditions in the region, 2) small reduction in emissions associated with lower fuel usage and 3) an increased risk of higher N2O emissions.

Da Silva et al. (2014) compared soil structure and GHG gas production in the row and interrow positions under no-tillage (section 6.5 ii). In summary, there was no difference in N2O and CO2 production between the row and interrow positions, which may have been because pore continuity did not differ between these two positions. Holland (2004) assessed the environmental impacts of implementing conservation tillage practices. As part of the review, the impacts on GHG (mainly CO2) emissions was also considered.

d) Impacts of bioenergy crops

Two studies, Brando et al. (2004) and Hiller et al. (2009) investigated the impact of bioenergy crops on GHG emissions (section 6.5 i).

e) Impacts of other management strategies or general review papers

Ball et al., (2002) investigated the impact of land use and land use change on soil GHG fluxes; Falloon et al., (2004) modelled N2O fluxes following the implementation of different field margin management strategies; and a number of researchers have reviewed the impact of land management on GHG emissions, including Garnett (2011), Friebauer et al., (2004) and Smith et al., (2013b).

Policy and technological constraints to GHG mitigation in agriculture were reviewed by Smith et al. (2007), while, Snyder et al. (2014) highlighted that policy should focus upon improved N-use efficiency to lower N2O emissions per unit of product or per unit of land area.

Antille et al. (2015) reviewed the potential for controlled traffic farming (CTF) to reduce GHG emissions. In summary, it was reported that both circumstantial and direct evidence indicates that CTF can reduce N2O emissions by 20% - 50% compared with non-CTF. However, if not managed correctly, there is an elevated risk of increased N2O emissions from the permanent traffic lanes. The improvement and/or maintenance of field drainage has been identified as a potentially significant mitigation method to reduce direct N2O emissions (Dobbie and Smith, 2006; Moran et al., 2008; MacLeod et al., 2010; Newell Price et al., 2011; Rees et al., 2013), although the magnitude of abatement and the agricultural area to which this may apply in the UK remains uncertain (Harris et al., 2009; MacLeod et al., 2010 (section 7.4). Rees et al. (2013) reviewed a number of mitigation strategies to reduce N2O emissions from agricultural soils, the measures proposed largely involved improving the efficiency of N fertiliser use and soil conditions.

Working with farmers

In addition to research projects on soil management practices, a number of initiatives involve farmer engagement and opportunities for farmers and researchers to work together.
to improve soil management practices and provide information about their effectiveness in enhancing soil quality for production and environmental protection. Programmes or strategies for farmer engagement include the farming for a better climate research group (run by SRUC on behalf of the Scottish Government) and the AHDB monitor farms (http://cereals.ahdb.org.uk/get-involved/monitor-farms.aspx).

The Duchy Future Farming Programme (FFP) (Soil Association) funds farmer-based research to support sustainable innovation by British farmers. The Kellogg’s Origins Initiative funds research to help improve supply chain efficiency whilst meeting the sustainability demands of consumers. Stobart et al. (2015) presented the results from a farmer research programme funded by the Kellogg’s Origins Initiative, which examined how cover crops might best be used to improve farm performance whilst giving environmental benefits. The IBERS PROSOIL project was designed to link research in soil management with farm practice by inviting farmers to implement a soil management technique and measure the impact on soil health on their farm.

6.5.1 Implications for policy and further research

At each of the workshops, the importance of engagement between researchers and farmers was repeatedly emphasised as being a priority to ensure understanding of agricultural issues (e.g. practical and economic challenges) and to aid the effective implementation of new knowledge and technologies.

The scientific importance of long-term field experiments as highlighted in section 3 was recently discussed at the AAB conference (Northumberland, May 2015; Aspects of Applied Biology, vol 128, 2015). Examples of long-term experiments include: the Rothamsted Research experimental platforms at Harpenden and North Wyke; the James Hutton Institute Centre of Sustainable Cropping (Perthshire); NIAB-TAG sites in Suffolk (STAR) and Norfolk (New Farm Systems), which investigate rotation and cultivation systems; and ADAS experimental platforms including field drainage sites at Boxworth (Cambridgeshire) and Faringdon (Oxfordshire) and long-term organic manure sites at Bridgets (Hampshire), Gleadthope (Nottinghamshire) and Terrington (Norfolk).

Recommendations for future research include further work to:

- Fully understand the response of microbial communities to disturbance and to assess the relationship between microbial diversity and function (Anderson et al., 2013).
- Improve our understanding of N₂O emissions in response to climatic and management drivers (Rees et al., 2013). Furthermore, it is unclear the extent to which N₂O mitigation strategies contribute to pollution swapping.
- Develop evidence-based practices to meet the challenges of ‘sustainable intensification’ to support policies and practices required to improve soil management practices (Powlson et al., 2011).
- Develop techniques to maintain food production with decreasing water resources (Powlson et al., 2011). Initiate and maintain long-term field experiments, in a range of locations to investigate critical soil P and K concentrations required to maximise crop yields (Powlson et al., 2011).
• Close nutrient cycles to deliver maximum yields with minimum losses to the environment (Dungait et al., 2012). Research will be required to fill our knowledge gaps on how nutrient management affects C, N and P cycles.

In addition, through discussions at the workshops it was highlighted that:
• Research into no-tillage practices within the UK is extremely limited, further work is required to fully understand the impacts of this cultivation method in UK agro-climate conditions.
• Evidence for improvements in soil quality and function, due to the implementation of agri-environmental schemes should be collated in order to assess their effectiveness.

6.6 How do different soil types respond to management practices and do they still have capacity to store more soil C?

Volume and characteristics of the overall evidence base

The literature research and screening process resulted in the identification of 23 papers related to how different soil types respond to management and whether soils have capacity to store more C. Two papers investigated the response of specific soil types (humid sandy loams and marginal sandy soils) to land use and management (Abdollahi et al., 2015; Tye et al., 2013); one paper looked at the effect of soil management (grazing exclusion) on SOC levels (Medina-Roldan et al., 2012); and another paper investigated the effect of soil type on soil quality in terms of total and active bacterial communities (Girvan et al., 2003). Thirteen papers investigated the capacity of soil to store more C (e.g. Dungait et al., 2012; Powlson et al., 2011; Smith et al., 2013, 2015; and Stockmann et al., 2013); four presented models for estimating SOC content (e.g. Meersmans et al., 2013; and Smith et al., 2010a, 2010b); and two were related to detecting change in SOC (Saby et al., 2008; Verheijen et al., 2005).

Defra project SP08016 also covered “Best Practice for Managing Soil Organic Matter (SOM) in (Lowland and Upland) Agriculture”; and SP1313 covered the “Assessment of the Effectiveness, Impact and Cost of Measures to Protect Soils”, including how different soil types respond to management and their applicability for different soil management practices.

What does the evidence base indicate in relation to the question posed?

Soil type response to land use and management

A number of papers allocated to section 6.5 addressed how different soil types respond to management. For example, Ball et al. (2007) assessed the impact of ploughing out grass/clover swards on soil physical fertility, soil structure and rooting conditions; Ball and Crawford (2009) investigated the effects of mechanical weeding on soil structure in light sandy and sandy loam soils; Guimaraes et al. (2013) related visual evaluation of soil structure to other physical properties in soils of contrasting texture and management; Gregory et al. (2009) reported on the effect of long-term soil management on the physical and biological resilience of a range of arable and grassland soils; Pheklan et al. (2013) investigated the physical effects of dairy cow treading on a clay loam soil; Douglas et al. (1998) identified structural improvements in a grassland soil following changes to wheel-traffic systems; and Clay et al. (2009) investigated the hydrological responses of upland...
blanket bog to managed burning and grazing. The searches carried out for this section resulted in an additional four papers:

Girvan et al. (2003) provided some useful information about the key factors determining total and active soil bacterial communities in the soils of three arable farms in eastern England. The results indicated that soil type rather than land management was the key factor determining bacterial community composition in these arable soils. The active bacterial communities were further discriminated by farm location and land use practices. Leguminous crops on particular soil types had a positive effect on SOC levels and resulted in small changes to the active bacterial population. The active (rather than the total) bacterial population was therefore more indicative of short-term management changes.

Tye et al. (2013) investigated the response of sandy soils formed from the Sherwood Sandstone in Nottinghamshire, England to changes in land use since the late 18th century. They found that the sandy soils under woodland had low concentrations of base cations, an acid pH and a mean SOC concentration (0-15 cm) of 2.7%. By contrast, arable soils had high base saturation percentage, pH close to neutral and mean SOC concentration (0-15 cm) of 1.7%. These results indicate the relatively rapid change in sandy soil properties that can result from land use change.

In an assessment of the response to management of humid sandy soils in Demark, Abdollahi et al. (2015) found that a winter cereal dominated crop rotation resulted in the poorest soil structural quality and produced the lowest relative yield compared with mixed rotations; and that mouldboard ploughing resulted in the best soil structural quality, and consequently the highest relative crop yield compared with reduced tillage treatments. Significant correlations were found in most cases between soil quality indices (including the Visual Evaluation of Soil Structure - VESS) and relative yield.

Medina-Roldan et al. (2012) looked at the effect of 7 years of grazing exclusion on vegetation composition and associated impacts on soil properties including soil C storage in an upland acidic grassland soil in northern England. The findings indicated that grazing-exclusion resulted in a slowing down of C and N cycling rates. However, this had no detectable impact on total C and N stocks in surface soil within the time frame of the study. While increases in soil C and N stocks might be expected in the longer term, the results suggest that a certain level of grazing is compatible with the provision of ecosystem services such as soil C storage under upland grassland management.

There was not a wealth of papers discussing how different broad soil types (e.g. mineral, organo-mineral and peaty) respond to soil management practices although Defra project SP08016 covers this in terms of changes in SOC (a fundamental soil property for soil resilience and function) in lowland and upland soils.

Soil carbon (C) storage
Globally, soils contain about 1,500 Pg (1Pg = 1 Gt = 10^5 g) of organic C, about twice that found as CO2 in the atmosphere and three times the amount of C in vegetation (Batjes, 1996; IPCC WGI, 2001). However, soil C pools are smaller now than they were prior to human intervention. Globally, soils have lost between 40 and 90 Pg C through deforestation,
cultivation and other forms of disturbance and soil degradation (Lal, 1999). Nevertheless, there is potential for some soils to store more C and a possible role for soil C sequestration in climate change mitigation (Smith et al., 2015). Whether ‘natural’ SOC levels will increase or decrease under future climate change scenarios is highly uncertain (see section 5.4). Research efforts should therefore be focused on the land management practices that can protect and enhance SOC levels (Smith et al., 2008).

Land management practices that can potentially contribute towards soil C storage include land use change (e.g. conversion of arable land to grassland or forest), re-vegetation of degraded land, improved plant productivity (due to soil manipulation and crop selection/breeding; and not simply from the use of manufactured N fertiliser; Powlson et al., 2011), stabilisation of C in subsoil, residue management, more effective use of organic materials (e.g. compost), optimal livestock densities and the use of herbs and legumes in grassland (Smith et al., 2008).

Such measures in combination could potentially have an estimated technical potential to increase SOC stocks by about 1-1.3 Pg a⁻¹ (Lal, 2004). However, to date estimates of the stocks and flows of C at the global scale to 2030, based on local knowledge of likely changes in land use and management, have predicted declines in C stocks in the Brazilian Amazon and Kenya, no change in the Indo-Gangetic Plains in India and a small increase in Jordan under one of three modelling approaches used (Milne et al., 2007). Furthermore, there are significant constraints that will limit the potential for C sequestration through land management and if not taken into account can result in an overestimation of the potential for C storage in soils. These include: (i) saturation of the C sink – the quantity of soil C that can be stored is finite as a new equilibrium is reached (Johnston et al., 2009), (ii) non-permanence – the process is reversible, which is particularly likely in arable systems (Freibauer et al., 2004), (iii) leakage/displacement – arable reversion to grassland or forestry leading to conversion of land from native vegetation to agriculture elsewhere in the world (e.g. Searchinger et al., 2008).

Adding organic materials to soils, while increasing SOC, generally does not constitute an additional transfer of C from the atmosphere to land, depending on the alternative fate of the residue (Powlson et al., 2011). For example, adding farmyard manure to the land does not constitute additional C transfer since most livestock manures are already recycled to land. However, recycling of compost to land that would otherwise be landfilled does constitute a genuine contribution to climate change mitigation (Bhogal et al., 2009). Furthermore, any measure that increases SOC is likely to have beneficial impacts on soil properties and functioning, including improved workability, water holding capacity, nutrient cycling and fertility (Blair et al., 2006; Watts et al., 2006). As such, soil C accumulation can be considered as a ‘no regrets’ policy for land management.

Increases in soil C from reduced tillage practices now appear to be much smaller than claimed in the past (Powlson et al., 2011, 2014). A large body of evidence indicates that SOC accumulation under no-till is relatively small, particularly in circumstances typical of temperate regions when soils under ‘no-till’ are cultivated conventionally (i.e. mouldboard ploughed) every few years to control weeds. Apparent increases in SOC are in large part due to altered depth distribution of soil C. Larger concentrations of C near the soil surface may
be beneficial for soil quality and crop growth and may help in adapting soils to climate change (see section 6.9), but its role in climate change mitigation has been widely overstated (Powlson et al., 2014).

Despite the limitations outlined above, there are significant benefits associated with increasing SOC concentrations. Under some agricultural management systems the current stock of C is below the optimal level for the delivery of multiple goods and services, and increasing SOC implies working towards levels that will allow all services to be delivered adequately (Nziguheba et al., 2015). One related long-term research goal with potential benefits for soil C sequestration is the development of perennial versions of current annual arable crops. This and other related possibilities were investigated in Defra project SP1605, sub-project A (2010) on “the potential of technologies for increasing C storage in soil to mitigate climate change”.

One proposed method of increasing C stocks in soils is to augment soil C pools with the longest mean residence times (MRT). However, Dungait et al. (2012) pointed out that the chemical composition of these pools and the capacity of decomposer organisms to move C between pools was not clearly understood. They discussed how advances in quantitative analytical techniques had redefined the new understanding of SOM dynamics and concluded that SOC turnover is governed more by accessibility of soil organisms to organic matter (i.e. disturbance/cultivation) than recalcitrance of the organic matter itself.

Modelling Soil C
Smith et al. (2010a) described the ECOSSE (Estimation of Carbon in Organic Soils-Sequestration and Emissions) model, which simulates soil C and N turnover in both mineral and organic soils using meteorological, land use and soil data available at the national scale. The modelled values showed a high degree of association with field measurements in both total C and change in C content of the soil. The model predicted that increasing the area of land use change from arable to grassland has the greatest potential to sequester soil C; and that losses were greater from C-rich soils (C content >6%; 64% of total soil C losses) than from non-C-rich mineral soils (Smith et al., 2010b). They suggested that two mitigation options that could be used in upland soils to achieve zero net loss of C from Scottish soils was to increase the amount of semi-natural land relative to improved grassland.

Meersmans et al. (2013) recognised that there were few plant residue C input values available and that this constituted a significant limitation to the C input parameters of current SOC stock simulation models. They estimated plant-derived soil C input values for agricultural sites in France based on data from 700 sites from the French soil monitoring network (the RMQS database). Measured SOC stock values were used to constrain an inverse RothC modelling approach to derive estimated C input values consistent with the stocks. The study offered an approach to meet the urgent need for crop-specific and environment-specific C input values in order to improve the reliability of SOC stock prediction.

Monitoring Soil C
Saby et al. (2008) assessed the feasibility of verifying the effects of changes in land use or management practice on SOC by comparing minimum detectable changes in SOC
concentration for existing European networks. Considerable effort would be necessary for some member states to reach acceptable levels of minimum detectable change for C sequestration accounting. Therefore, it is unlikely that soil monitoring networks could be used for annual national C accounting, but the importance of organic C for the delivery of soil functions underlines the importance of establishing effective national soil monitoring networks.

6.6.1 Implications for policy and further research

It is important that land management policies take account of the C sequestration potential of different soil types and land management practices in UK agro-climatic conditions while also recognising that policies encouraging the protection or enhancement of SOC represent a ‘win-win’ strategy in terms of climate change mitigation and the sustainability of agricultural systems. This also underlines the importance of supporting effective national soil monitoring networks for the assessment of soil C concentrations and stocks.

Further work is needed to:

- Update existing soil C models to take account of a better understanding of SOM dynamics.
- Produce crop-specific and environment-specific C input values to improve the reliability of SOC stock predictions.
- Collate a substantial body of research in order to assess how different broad soil types (e.g. mineral, organo-mineral and peaty) respond to soil management practices within arable and grassland systems.

6.7 Can soil biodiversity be manipulated to improve ecosystem service delivery?

Volume and characteristics of the overall evidence base

The literature review and screening process resulted in the allocation of eight papers to this question. Of the eight papers identified, two contributed to the understanding of soil biodiversity in general (Usher et al., 2006; Wall et al., 2010); two investigated the feasibility of enhancing grassland biodiversity for ecosystem delivery (De Deyn et al., 2011; Smith et al., 2008); one focused on the role of soil microbes in the global C cycle (Gougoulias et al., 2014); one examined the interactions between earthworms, fungi and soil structural properties (Spurgeon et al., 2013); one investigated the impact of arable systems on biodiversity and soil function (Stoate et al., 2001); and one discussed “novel biological approaches” to mitigate global environmental change (Woodward et al., 2009). Four of the papers focused on manipulating soil biodiversity for climate change mitigation; two on improving the reliability and resilience of food production; and one on water regulation.

What does the evidence base indicate in relation to the question posed?

Soil biodiversity

Usher et al. (2006) described the origins, development and characteristics of a major soil biodiversity research programme, the NERC Thematic Research Programme ’Biological Diversity and Function in Soils’. The programme had six scientific aims, among which were to manipulate major taxonomic groups of soil biota under controlled conditions to determine the extent to which soil biodiversity is an indicator of soil ecosystem resilience. The paper
outlined the rationale for the study at a single, upland grassland site, and provided results of the soil chemistry and botanical composition background monitoring. The main achievements of the programme were an improved understanding of the high level of diversity of soil bacteria, protozoa, mycorrhizal fungi and nematodes, and the speed with which processes such as C cycling occur in the soil. The research constituted a significant advance towards understanding both the extent and function of soil biological diversity, but for a single upland site only.

Wall et al. (2010) described how a multitude of soil organisms contribute toward supporting food and fibre production, decomposition and nutrient cycling. However, they stress how relating changes in soil biodiversity to shifts in ecosystem function is extremely challenging.

Management practices that influence earthworm numbers were explored as part of the PRO-SOIL project. At the IBERS experimental site, digestate application over a 4-year period, significantly increased earthworm numbers compared to the application of manufactured NPK fertiliser. At Cappele (North Wales) slurry application, over a 4-year period, was also found to increase earthworm numbers compared to no slurry. At IBERS, from spring 2010 to autumn 2014, there was no difference in earthworm abundance between aerated and non-aerated slurry. The impact of different forage crops on earthworm numbers was also tested; earthworm numbers were significantly greater in soil under white clover compared to perennial ryegrass.

Enhancing grassland diversity for ES delivery
De Deyn et al. (2011) found that long-term biodiversity restoration practices increased soil C and N storage especially when these treatments were combined with the introduction and promotion of red clover (Trifolium pratense). Long-term diversity restoration practices can yield significant benefits for soil C storage when they are combined with increased abundance of a single, sub-ordinate legume species in low to moderate output grassland systems. De Vries et al. (2013) also demonstrated how manipulating plant and soil biodiversity can enhance C and nutrient cycling and improve productivity in low output grassland systems.

Smith et al. (2008) assessed the effect of seed mix and management on the biodiversity, conservation and functional value of grassland field margins for soil macrofauna. Diversity in the grass margins was generally higher than in the crop, with Lumbricidae, Isopoda and Coleoptera having significantly more species and/or higher abundances in the margins. However, scarification of the margins reduced the diversity and abundances of Isopods; and soil- and litter-feeder abundances and predator species densities. The findings indicated that appropriate management of grass margins, including spring cutting and spot herbicide application to control invasive weeds (but not scarification) can significantly enhance their value for soil macrofauna. Minimising cultivation and developing a substantial litter layer in grass margins can encourage litter-dwelling invertebrates that tend to be missing from arable systems. However, this may conflict with other aims of agri-environment schemes, such as enhancing floristic and pollinator diversity.
Macro-fauna and soil quality
Spurgeon et al. (2013) provided a quantitative assessment of the effects of land use and land management changes on community metrics across a range of successional transitions (conventional arable; no or reduced tillage; grassland; and wooded land) for two functionally important soil taxa, earthworms and fungi. The findings indicated a consistent trend of increased earthworm and fungal community abundances and complexity following transitions to lower intensity and later successional land uses. The greatest changes were seen for early stage transitions, such as introduction of reduced tillage regimes and conversion to grassland from arable land. Alterations in soil communities were associated with changes in soil structure and hydrology. For example, fungal biomass was positively associated with soil micro-aggregate stability; and earthworm abundance and functional group composition were positively correlated with water infiltration rate (dependent on tillage regime and habitat characteristics).

Micro-fauna and C cycling
Gougoulias et al. (2014) discussed the role of soil microbes in the global C cycle and reviewed the main methods that have been used to identify the microorganisms responsible for the processing of plant photosynthetic C inputs to soil. They discussed whether application of these techniques could provide the information to underpin the management of agro-ecosystems for C sequestration and increased agricultural sustainability. They concluded that current technologies lack the ability to provide sufficient quantitative information on how C is apportioned between different microbial functional groups to inform rational manipulation of the plant-soil system and favour organisms that promote soil C storage in agricultural soil.

Soil biodiversity and soil function in arable and grassland systems
Stoate et al. (2001) argued that the development of high input arable systems and simplified crop rotations has been associated with a decline in biodiversity, the loss of non-crop habitats and simplification of plant and animal communities within crops, with consequent disruption to food chains and declines in many farmland species. They stressed that there is still an opportunity to implement measures to improve soil and plant biodiversity through a combination of cross-compliance and agri-environment schemes that could result in more efficient delivery of key ecosystem services.

Boag et al. (2005), assessed the distribution of the New Zealand flatworm, an alien species to the UK and a predator of native earthworms. Further research has demonstrated that at densities comparable to natural infestations within grasslands the New Zealand flatworm has the capacity to reduce earthworm biomass by 20%, which may in turn negatively impact on soil functioning.

6.6.1 Implications for policy and further research
The evidence indicates that more information is needed to determine the key drivers that determine the effective delivery of ES in agricultural landscapes and the degree to which certain biological assemblages are essential to sustainable soil systems. A better understanding of how soil biodiversity can be manipulated and indeed how soil management practices influence soil biodiversity at different spatial and temporal scales...
(e.g. the impact of specific seed mixes and mechanical interventions on earthworm biomass and number) would help guide soil and land management policies.

The collated research indicates a need to:

- Provide more quantitative information on how C is apportioned between different microbial functional groups to inform whether or not the plant-soil system can be manipulated to favour organisms that promote soil C storage in agricultural soils.
- Assess the C sequestration potential of manipulating plant species diversity and associated soil biodiversity in grassland soils and implications for overall food production levels.
- Carry out pilot-level field work, to examine the feasibility of, and reduce the uncertainties in, biological mitigation strategies.
- Engage with the public concerning mitigation strategies as certain biological approaches may have negative effects on some ecosystem services and land use (Woodward et al. 2009).

6.8 How may climate change affect the choice of soil management measures to achieve sustainability?

Volume and characteristics of the overall evidence base

Twenty five papers provided information on how climate change may affect land management. Ten papers investigated how agriculture in general may have to adapt to climate change (e.g. Blackmore, 2014; Rivington et al., 2013; Vermeulen et al., 2012); and seven papers provided information on adaptation strategies within specific agricultural systems (e.g. Hopkins and Del Prado, 2007; House et al., 2010; Orr et al., 2008). None of the papers explicitly dealt with how climate change may affect the choice of soil management measures.

Two papers discussed the effect of climate change on crop yields (Deryng et al. 2011; Lee et al., 2010). Two papers investigated how climate change may influence DOC concentrations in peat upland soils (Evans et al., 2012; Yallop et al., 2010); one paper investigated changes in land suitability for potato production (Daccache et al., 2012); another discussed the effect of climate change on macronutrient cycles (Whitehead and Crossman, 2012); and one discussed adaptation to climate change in non-agricultural systems (Pritchard et al., 2014).

A number of Defra projects have provided information on how agriculture may have to adapt to climate change. These include Defra projects:

- SP1316 (2015) on “identifying the soil protection benefits and impact on productivity provided by the access to waterlogged land requirements in cross compliance and exploring the impacts of prolonged waterlogging on soil quality and productivity”;
- AC0302 (2009) on “a research and innovation network supporting adaptation in agriculture to Climate Change”;
- AC0308 (2006) on “ecosystem services for climate change adaptation in land management”; and
- CA0511 (2012) on the “role of payments for ecosystems services in climate change adaptation”.

109
What does the evidence base indicate in relation to the question posed?

Adaptation in agriculture

Ability to adapt to changes in climate assumes to some extent that such environmental change can be predicted with some certainty. Falloon et al. (2010) and Rivington et al. (2013) stressed the high level of uncertainty in climate change projections due to complex interactions between the atmosphere, biosphere and hydrological cycle and emphasised the need for more integrated approaches to climate impact assessments.

Smith and Olesen (2010) describe how climate change could involve increases in the variability of temperature and rainfall and that adapting to these conditions would require greater resilience in agricultural production systems. This may be achieved through improving soil water holding capacities, by adding crop residues and manure to arable soils or by adding diversity to the crop rotations. They argue that most adaptation options have positive impacts on climate change mitigation. These include:

i. measures that reduce soil erosion
ii. measures that reduce nitrate leaching losses
iii. measures for soil moisture conservation
iv. increasing the diversity of crop rotations
v. modification of microclimate to reduce temperature extremes and provide shelter for livestock
vi. land use change involving abandonment or extensification of existing agricultural land, or avoidance of the cultivation of new land

They conclude that these adaptation measures will in general, if properly applied, reduce GHG emissions, by improving N use efficiencies and improving soil C storage.

Rivington et al. (2013) described the use of agro-meteorological metrics (i.e. indicators of weather-determined environmental conditions), derived from observed weather and Regional Climate Model projection data for 12 sites in Scotland, to estimate future changes in agricultural management. The results indicated projected changes to seasonal rainfall distribution (drier soil conditions in autumn and wetter soils in spring), growing season length, soil moisture deficits and accessibility; and a potential need for substantial changes in agricultural systems and land use. Potential changes included warmer conditions and earlier (and later) grass growth, which could increase production potential with implications for soil N balances, especially if growth occurs while the soil is moist in spring and autumn. Potential restricted water availability may have implications for irrigation and SOC strategies and a shift to drier soil conditions in autumn and wetter soils in spring may alter crop rotation choices (i.e. more winter cropping).

Schnug et al. (2008) investigated potential synergies between agriculture and nature conservation under projected climate change scenarios. They argued that an increase in productivity and longer growing seasons in northern Europe (Deryng et al., 2011) could justify higher manufactured fertiliser input and that an increase in the use of legumes and diversification of crop rotations could counteract the potential for increased nutrient losses, while also being beneficial for biodiversity. Actions to improve soil structure and SOC
content could also improve water infiltration thereby enhancing water conservation for summer crop growth and reducing flooding risk.

Shepherd et al. (2012) examined thirty mathematical models for their ability to evaluate the impact of proposed agricultural management practices on the environment under future climate change scenarios. The models were assessed against a set of four essential criteria:

i. spatial and temporal scale, ease of use, and ability to consider a change in climate;
ii. ability to simulate nutrient cycling processes, specifically C and N dynamics with microbial turnover, mineralization-immobilization, nitrification and denitrification, plant nutrient uptake, and phosphorus cycling;
iii. ability to consider a water balance and water movement through soil; and
iv. ability to introduce and modify agricultural practices relating to crop and livestock management.

Shepherd et al. (2012) concluded that no single model met all the stated criteria, but that three models accommodated most features and could be developed to provide a general assessment of the effects of farm mitigation and adaptation on environmental losses under a changing climate.

Falloon et al. (2010) focused on the impact of projected hydrological changes, potential increases in crop productivity (Lee et al., 2010) and an increase in extreme rainfall events and droughts on agricultural adaptation options. They anticipated that increased flooding and waterlogging risk (mainly in winter) and summer irrigation shortages may present challenges for agriculture. Although the need for effective adaptation could be greatest in southern Europe, all regions should benefit from strategies to increase SOC levels to improve the resistance and resilience of soils to extreme weather events.

Prolonged waterlogging can be a significant constraint on livestock farming and tillage systems (Schulte et al., 2010). Defra project SP1316 considered changes to farming practices that may be required to adapt to warmer, wetter winters and protect against a more frequent occurrence of prolonged waterlogged conditions. It was concluded that farming adaptations that improve soil drainage, maintain or enhance organic matter levels and produce better structured soils will help to mitigate against waterlogging and will generally benefit the farm business, although economies of scale are an important consideration and significant changes to cultivation systems, such as the adoption of controlled traffic farming (CTF) can have practicality and cost limitations. Adaptations that could contribute to improving farm profitability and make farm systems more resilient to climate change included:

- Replacing and maintaining field drainage systems on slowly permeable soils
- Applying bulky organic manures to improve soil resistance and resilience
- Avoidance of compaction, including the use of low ground pressure tyres in suitable field conditions
- Visual assessment of soils combined with timely use of sub-soilers and top-soilers, where appropriate
The use of more efficient machinery to establish crops more rapidly, particularly on larger farms.

Climate change adaptation should also consider socio-economic aspects of change (Fazey et al., 2010; Blackmore, 2014). Fazey et al. (2010) presented a conceptual framework for adaptation with a view to developing more effective long-term strategies that incorporate technological fixes as well as addressing human behavioural causes/change and the need to enhance human skills conducive to sustainable management. This is supported by Blackmore (2014), who considered the type of social infrastructure that will need to be in place to support learning for adaptation. They drew key examples from two research contexts, one concerned with innovations in agricultural machinery and the other with water management and governance.

Rivington et al. (2007) argued that an integrated assessment approach, combining simulation modelling with deliberative processes involving decision makers and other stakeholders, has the potential to generate relevant assessments of how climate change may impact on farming systems and the kinds of resilience and adaptive capacity that will need to be developed. Vermeulen et al. (2012) presented a summary of current knowledge on options to support farmers in achieving food security under climate change. They outlined two broad overlapping areas for adaptation: (i) accelerated adaptation involving integrated packages of technology and agronomy, and (ii) better management of agricultural risks associated with increasing weather variability and extreme events, for example improved climate information services.

Adaptation in grassland, peatland and upland systems

Hopkins and Del Prado (2007) identified adaption strategies in grassland systems as a response to projected climate change that is likely to increase herbage growth (Lee et al., 2010) and to favour legumes more than grasses, but limit moisture supply in some (mainly eastern) areas of the UK. Further implications for grasslands may arise from increased frequency of droughts, storms and other extreme events (Hopkins and Wilkins, 2006).

Fry et al. (2014) investigated the effects of extreme changes in rainfall regimes on ecosystem functioning in a mesotrophic grassland system. They found that soil extractable P and ecosystem respiration were significantly higher in rainfall change treatments, but that on the whole ecosystem functioning in the grassland soil studied was resistant, in the short term, to extreme rainfall changes, although they did stress that prolonged study is needed to measure longer-term impacts.

House et al. (2010) explored the impacts of climate change in the British Uplands. They reported that under low and high emission climate change scenarios, as much as 50% of British Uplands and peatlands will be exposed to pressures from erosion and potential C loss by the end of the 21st century, although process-based model projections are highly uncertain. Nevertheless, they conclude that preserving upland vegetation cover is a key ‘win-win’ management strategy that will reduce erosion and loss of soil C, and protect a variety of services such as the continued delivery of a high quality water resource.
Orr et al. (2008) suggested that land management policies should be modified to secure soil C stocks, and to protect the other goods, services and functions that uplands deliver. Potential upland adaptation strategies include improved spatial planning for land and water, the creation of networked habitats to enable species migration, and practical guidance on appropriate locations for intensification and extensification of land use. Joosten et al. (2015) promoted a production system known as paludiculture for upland and lowland peat soils. Paludiculture involves the use of biomass from wet and rewetted peatlands under conditions that maintain the peat body, facilitate peat accumulation and continue to provide provisioning and regulating ecosystem services. Examples include low-intensity grazing with water buffalo, biofuels from fens, common reed as industrial raw material and sphagnum farming for horticultural growing media.

Climate-Smart Grass (http://www.nrn-lcee.ac.uk/climate-smart-grass/) is an on-going research project funded by the Welsh National Research Network for Low Carbon Energy and the Environment (NRN-LCEE). Climate-Smart Grass investigates plant-soil interaction in grasslands and assesses the effects of extreme weather events (i.e. flooding, drought and high ground-level ozone levels) on productivity and the delivery of ecosystem services. The aim is to propose management regimes that will improve the resilience of farming to environmental stresses. Furthermore the project aims to determine ‘tipping points’ at which environmental stressors cause irreversible damage to grassland ecosystem functioning.

6.8.1 Implications for policy and further research

Adaptation strategies that increase the resistance and resilience of agricultural soils, such as maintaining field drainage systems on slowly permeable soils, growing cover crops, applying bulky organic manures and avoiding soil compaction also generally help improve farm profitability and should be encouraged through policy.

The evidence supports the need for:

- Better quantification of C fluxes under different soils and land management practices.
- Techniques for up-scaling local interventions to quantify landscape-scale benefits.
- More prolonged study of the potential impacts of changes in temperature and seasonal rainfall patterns (under a range of potential climate change scenarios) on whole ecosystem functioning in grassland and arable soils to provide a better understanding of longer-term impacts.
- The development of more comprehensive process-based models combining mathematical relationships and expert opinion to provide a general assessment of the effects of farm mitigation and adaptation on environmental losses under a changing climate.
- Investigation into the social, economic and environmental viability of new climate change mitigation and adaptation techniques such as paludiculture.
- More integrated approaches to climate impact assessments including consideration of socio-economic aspects of climate change adaptation.
7. Economic value of ecosystem services

7.1 Introduction
It has become increasingly important in policy and decision-making circles to consider the 
economic benefits (in addition to aesthetic value) that are gained from well-functioning 
ecosystems. This section focuses on the economic value of the ecosystem services provided 
by soil. It presents the ecosystem services that soils provide before considering how these 
can be valued economically. Papers that have attempted to put an economic value on soil ES 
or provide information that would inform such an evaluation are summarised. Other papers 
consider how soil ES can be valued and report on the lack of decision support tools that can 
adequately assess soil functionality and ES delivery. A few papers cover how soil 
management relates to soil functionality to help quantify the value of soil ES, but this is also 
recognised as a fundamental limitation of current decision support tools. Finally, a synthesis 
of papers that assess the relative cost and societal benefit of soil management practices is 
provided.

A total of 29 papers were identified as relevant to the valuation of ES provided by soils 
(Figure 13).

Figure 13. Number of papers identified as relevant to addressing specific questions within 
the “economic value of ecosystem services” theme.
7.2 What is the economic value of the key ecosystem services provided by sustainably managed soils?

Volume and characteristics of the overall evidence base

The search and screening process resulted in 29 research papers that were related to the economic valuation of ecosystem services. However, these 29 papers reference many more related documents and sources. Of the 29 papers identified, ten covered the economic evaluation of ES in general (i.e. valuing ‘nature’); 11 investigated the evaluation of ES provided by soils or specific soil types (e.g. peat); two considered the economic value of biodiversity; and three specifically covered the value of the agricultural sector in terms of ES (Table 19).

Table 19. Themes covered by research papers investigating the economic valuation of ecosystem services.

<table>
<thead>
<tr>
<th>Economic valuation of ecosystem services - themes covered</th>
</tr>
</thead>
<tbody>
<tr>
<td>General</td>
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<tr>
<td>---------</td>
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<tr>
<td>10</td>
</tr>
</tbody>
</table>

What does the evidence base indicate in relation to the question posed?

“Value” is “that quality of an object that permits measurability and therefore comparability” (Robertson, 2012). This section covers economic value rather than a wider meaning of value incorporating “normative and moral frameworks people use to assign importance and necessity to their beliefs and actions” (Farber et al., 2002). Value can be “extrinsic”, which is when a service or action is valued by its function, or “intrinsic”, which is when an object is valued in itself as is the case for aesthetic or moral value (Robinson et al., 2014). Economic value is based largely on an anthropocentric, extrinsic view. Total economic value (TEC) is the sum of all relevant use and non-use, market and non-market values provided now or in the future (i.e. option value) by a good or service (Costanza et al., 1997).

Total economic value requires an estimation of market and non-market values. There are long standing disagreements about the meaning of non-market estimates (Vatn, 2004), and a whole range of moral, ethical and philosophical criticisms that have been made against non-market evaluation (e.g. Sagoff, 1988). Robinson et al. (2014) note that, given the limitations and criticisms of valuation for policy, the real merit in conducting this type of exercise is not necessarily the “number” that emerges but more the process that is undertaken and the need to consider how society can benefit from a particular soil process or function.

The literature indicates that we are not yet in a position to assign an economic value to the ecosystem services that soils provide. Valuing soil comprehensively presents challenges. Fraser and Robinson (2015) identify a need to take account of the direct value of soil in accounting frameworks such as the UN System of Environmental Economic Accounting (http://unstats.un.org/unsd/envaccounting/pubs.asp), to differentiate between the value of soil and the value of land, and to provide more scientific evidence of the impact of management interventions on ‘soil change’ and ecosystem function. We need a better understanding of the trade-offs between what we can use soils for and what they provide.
and a more complete valuation of soils (Fraser and Robinson, 2015). Nevertheless, a number of papers consider how soils can be valued and summarise the work that has been done on valuing soil ecosystem services.

Costanza et al. (1997) estimated the economic value of 17 ecosystem services, based on published studies and a few original calculations. For the entire global biosphere, the value (most of which was non-market value) was estimated to be in the range of US$16–54 trillion (10^{12}) per year, with an average of US$33 trillion per year; compared with total global gross national product (in 1997) of around US$18 trillion per year. Costanza et al. (1997) do not break down the value of the total biosphere into the ecosystem services provided by soil and question the usefulness of doing so. Nevertheless, their assessment provides an indication that this should be possible. They state that “It is not very meaningful to ask the total value of natural capital to human welfare, nor to ask the value of massive, particular forms of natural capital. It is trivial to ask what the value of the atmosphere is to human kind, or what the value of soil infrastructure is as a support system. Their value is infinite in total. However, it is meaningful to ask how changes in the quantity or quality of various types of natural capital and ecosystem services may have an impact on human welfare”.

Many of the papers did not target soil ecosystem services specifically and covered the issue of whether it is still valid, appropriate, useful or even possible to put a monetary value on ecosystems. A number of papers have helped to identify valuable soil functions and ES as a first step in the process of valuing soils (e.g. Daily et al., 1997; Lavelle et al., 2006; Haygarth and Ritz, 2009; Dominati et al., 2010; Robinson et al., 2012), including provisioning regulating and cultural services (Table 20). However, there are few key papers that discuss how soils can be valued and even fewer that provide a set or range of economic values for any particular ES.

Dominati et al. (2010) developed a framework to classify and quantify soil natural capital and ecosystem services. The framework consisted of five main interconnected components:

i. soil natural capital, characterised by standard soil properties well known to soil scientists;
ii. the processes behind soil natural capital formation, maintenance and degradation;
iii. drivers (anthropogenic and natural) of soil processes;
iv. provisioning, regulating and cultural ecosystem services; and
v. human needs fulfilled by soil ecosystem services.

Fisher et al. (2008) discussed key concepts involved in formalizing ecosystem service research. These included the distinction between services and benefits, understanding the importance of marginal ecosystem changes, formalizing the idea of a ‘safe minimum standard’ for ecosystem service provision, and discussing how to capture the public benefits of ecosystem services.

Defra project SP1606 aimed to estimate the total economic cost of soil degradation in England and Wales in order to inform priority areas for future research and policy. Six main processes of soil degradation and their effects on soil quality were identified for analysis, namely: erosion, compaction, decline in organic content, loss of soil biota, diffuse
contamination and surface sealing. Soil degradation costs ranged between £0.9 bn and £1.4 bn per year, with a central estimate of £1.2 bn. About 45% of total quantified annual soil degradation costs were associated with loss of soil organic matter, 39% with compaction and 13% with erosion. Around 20% of the estimated annual costs of soil degradation were associated with loss of provisioning services linked with agricultural production. The remaining 80% of total annual degradation costs were associated with loss of regulating services, the bulk of this (49% of all costs) linked to GHG emissions.

In Scotland, Dobbie et al. (2011) carried out an evaluation of the relative importance of the threats to soil functions using assessment criteria and a scoring system developed for the project. This assessment provided new evidence to enable future prioritisation of resources and focus activities on the most important issues for Scotland’s soils. In addition, a socio-economic assessment of the impacts of soil threats was derived from information produced by Glenk et al. (2010).

At the scale of the soil organism, attempts have been made to value the ecosystem services provided by earthworms. For example, the PROSOIL project (PROSOIL, 2014) estimated the financial value of earthworms in terms of the topsoil they produce as £3 to £10.50 ha\(^{-1}\) yr\(^{-1}\), based on assumptions developed by Wratten et al. (2011).

Robinson et al. (2013) proposed that an appropriate ecosystems framework for soils should incorporate soil stocks (natural capital) showing their contribution to stock-flows and emergent fund-services as part of the supply chain. In the Millennium Ecosystem Assessment soils were given the vital role of a supporting service, but many of the other soil goods and services remained obscured. They demonstrated how an operational ecosystems concept for soils can draw on much more supporting data on soil stocks as demonstrated in a case study with soils data from England and Wales showing stocks, gaps in monitoring and drivers of change. The recent establishment of the Natural Capital Committee advising UK government is further evidence of a growing recognition of Natural Capital and of soil as an important component of natural capital (Fraser and Robinson, 2015).

Ekins (2011) proposed a new approach for environmental policy that goes beyond the cost-benefit analysis that has proved so difficult to implement for non-marginal environmental issues. The proposed approach combines the “Safe Minimum Standard” approach with the concepts of environmental functions and ecosystem goods and services. It provides the basis for an assessment of sustainability across different environmental themes, following which the ‘sustainability gap’, showing the extent to which the standard is not being met, can be quantified. The approach assesses sustainability performance in a scientifically robust, easily communicable indicator that may be compared with GDP.
Table 20. Soil goods and services and the types of value associated with them that make up the total economic value (source: Robinson et al., 2014).

<table>
<thead>
<tr>
<th>Total Economic Value (TEC)</th>
<th>Use value</th>
<th>Non-use value</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Direct &amp; marketable</td>
<td>Direct &amp; non-marketable</td>
</tr>
<tr>
<td>Provisioning services</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Topsoil</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Subsoil</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Peat</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Sand/Clay minerals</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Soil for rare earth extraction</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Soil organisms, earthworms Biomedical resources, antibiotics and new organisms used for medicine</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Provision of physical support</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Provision of food, wood and fibre</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Regulating services</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Waste processing</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Detoxification</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Nutrient recycling</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Nutrient/contaminant</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Filtering</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Water filtration</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Hydrological regulation</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>River flows mitigation/water levels</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Flood peak regulation</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Climate regulation</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>C storage</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Soil moisture buffering of heat and cold waves</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Greenhouse gases mitigation</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Hazard regulation</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Structural support</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>shrink-swell</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Dust emissions</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Liquefaction</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Landsliding and slumpng</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Pests and Disease regulation</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Human and animal pathogens</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Disease transmission and vector control</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Cultural services</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Burial ground</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Scenery</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Recreation</td>
<td>X</td>
<td>X</td>
</tr>
<tr>
<td>Preservation of artefacts</td>
<td>X</td>
<td>X</td>
</tr>
</tbody>
</table>
Chatterton et al. (2015) used an ecosystems-services-framework approach, and an integrated “livestock-ecosystems linear programming model” to assess economic and environmental impacts of the livestock sector. Analysis showed the main benefit coming from provisioning and cultural services, and trade-offs between these and the cost of regulating services with respect to emissions to air and water.

The valuation of soil ES has employed the full range of valuation methodologies (see Bateman et al. (2002) for the range of economic valuation methodologies). These include:

- The market value of soils and commodity prices – which can be regarded as a minimum value
- The cost of soil degradation – its effect on productivity and replacement cost
- Stated preference research – which can provide useful information for decision makers - e.g. value of soil conservation programs and the preference of farmers to adopt specific soil management practices

However, there have not been many instances of the use of these methodologies to value soil per se and few of them are particularly useful for valuing the ES that soils provide. The cost of soil erosion is the most commonly assessed aspect of soil ES valuation (e.g. Adhikari and Nadella, 2011); most commonly taking into account the on-site costs, but in some cases also the off-site costs borne by the rest of society (e.g. Pretty et al. 2000).

Dominati and Mackay (2013) used an ecosystem services approach to value soils on hill country beef and sheep farms in New Zealand. They focused on quantifying the costs of soil erosion and the value of soil conservation practices in terms of how they affected soil quality and the long-term provision of ecosystem services. The whole range of soil services were valued using a cost-benefit analysis.

As the importance of the multifunctional aspects of soils is increasingly recognised there will be an increasing need to develop decision support tools (DST’s) for assessing ecosystem services. Life cycle assessment (LCA) is one such methodology that is increasingly used in environmental impact assessment. Robinson et al. (2014) describe LCA as “a tool to quantitatively evaluate environmental impacts resulting from a product or service life cycle, from material extraction to waste management”. However, the accounting of ecosystem service provision in LCA presents some significant challenges (Bakshi and Small, 2011). These include the fact that regulating and other services are difficult to quantify in physical terms and that current LCA modelling does not represent the complexity and interaction of soil properties and the value of functions, such as nutrient cycling, mainly due to the difficulty of relating changes in soil quality to specific resource flows (Garrigues et al., 2012), which is a necessary step in LCA.

Robinson et al. (2014) state that there are no spatially explicit DST’s designed for soils or soil management. However, within the wider context of managing soils for multiple benefits, there are a number of tools developing (Bagstad et al., 2013). Examples include InVEST (Nelson et al, 2009), which uses soils data and predicts soil change as part of a full economics evaluation tool; the global TEEB (The Economics of Ecosystems and Biodiversity: http://www.teebweb.org/) initiative, which aims to incorporate ecosystem service values
into decision-making for policy makers (e.g. TEEB, 2011); and the Land Utilisation and Capability Indicator model (LUCI; Jackson et al., 2013). LUCI models a number of soil-mediated services including water regulation, climate change mitigation (C sequestration and storage) and crop production, and links to economic valuation using biophysical thresholds as part of a “trade-off evaluation component”.

However, in most spatial DST’s soils information is not incorporated in its own right and response functions linking the contribution of different soil types to ecosystem services are lacking. As the importance of the role of soils in delivering ecosystem services is increasingly recognised, these aspects within DST’s will need to be addressed.

It is recognised that ecosystem service concepts are not without criticism and a number of papers have questioned continued attempts to put a monetary value on ecosystem services, not least due to the challenge of explaining the concept to lay people and the associated difficulty of linking services to willingness to pay (e.g. Barkmann et al., 2008). For example, Admiraal et al. (2013) presented a framework for combining ecological theory with economic valuation, based on the premise that “total economic value bases the monetary value of ecosystems purely on the flow of human benefits of services of ecosystems and consequently ignores questions of sustainable use of natural capital per se”. The authors present theoretical ecological insights about ecosystem resilience which provides an economic perspective on investment in biodiversity. Total economic value is put in a framework where investment in biodiversity is expanded to cover functional diversity to maintain ecosystem resilience and promote sustainable use of ecosystems in something called “portfolio theory”.

Baveye et al. (2013) point out that “there is a long history, starting in the late fifties but largely ignored, of sustained attempts to assign monetary values to nature's services”. They argued that “the difficulties currently encountered in the practice of the monetary valuation of ecosystem services (MES) turn out to be long-standing problems that have eluded solution over the last half-century and appear intrinsically unresolvable”. They suggested that alternatives to MES should be sought to integrate nature into economic decisions.

One possible model is the payment for ecosystem services (PES), where government or other key stakeholders pay land managers to deliver services through good soil management. One such example, which did not focus on soils, but serves as a model for how PES could support good soil management practices, was the Fowey River Improvement Auction (Defra project NE0131), which was funded by South West Water to support capital investments on farms to improve water quality in the River Fowey of Cornwall. The project concluded that auction-based PES mechanisms are the best option for distributing funds when the benefits of investments can be estimated reasonably accurately without site-specific knowledge; while advisor-led PES schemes are to be preferred when an advisor’s expert judgement is needed on the ground, where the scale of the scheme is small and where advisors have good local knowledge with which to target farms likely to yield good investment opportunities.

Cornell (2010) stated that in applying ecosystem service concepts, we should remain alert to some ‘health warnings’ from previous experience in environmental economics:
The ecosystem services concept involves a narrowing of focus onto the monetary value of ecosystems.

There can be a conceptual disconnection of value from function, particularly when there is a reliance on benefits transfer.

Lack of data is a significant issue.

Firbank et al. (2013) reviewed the delivery of ecosystem services from enclosed agricultural land in the UK. They assessed the current state of the multiple services that agricultural land provides and argued that food production can be integrated into other services by promoting a diversity of agricultural systems and allocating land to the delivery of different goods and services according to its suitability. Approaches included using technology to minimise environmental impact; managing patches within fields or larger areas for environmental benefit; and developing markets and regulations for environmental protection.

Hodgson et al. (2007) explored the concept of 'ecosystem services' as a way of understanding how particular relations between nature and society can be established and studied and as a potential integrating approach for interdisciplinary research, but do not provide specific valuing methodologies.

Myers (1990) assessed several ecosystem services, including regulation of climate and biogeochemical cycles, hydrological functions, soil protection, crop pollination, pest control, recreation and ecotourism, and concluded that the services were significant in both ecological and economic terms. Ecosystem resilience was seen as particularly important in underpinning many of the services. The paper concludes with a brief overview assessment of economic values at issue and an appraisal of the implications for conservation planning.

Peatlands

Rawlins and Morris (2010) used a combination of stakeholder analysis and the functions, uses and values framework, to explore stakeholder preferences for peatland functions, along with the current drivers and associated responses in peatland areas. They found that non-use values such as nature conservation were more prevalent now than in the past, but also that there was renewed interest in sustaining the productive capacity of peatlands for some types of intensive farming. They identified an opportunity to develop a policy framework to encourage wise peatland management systems across Northern Europe by incorporating socio-economic factors and peatland ecosystems into peatland management decisions.

Peh et al. (2014) used a new Toolkit for Ecosystem Service Site-based Assessment (TESSA) to evaluate the changes in ecosystem service delivery resulting from a land conversion from drained, arable land to a wetland habitat mosaic at the Wicken Fen National Nature Reserve. The tool was used to estimate biophysical and monetary values of ecosystem services provided by the restored wetland mosaic compared with the former arable land. Restoration was associated with a net gain to society as a whole of $199 ha\(^{-1}\) yr\(^{-1}\) for a one-off investment in restoration of $2320 ha\(^{-1}\).
The loss from reduction in arable production was compensated for by estimated gains from:

- Nature-based recreation
- Grazing
- Flood protection
- Reduced GHG emissions
- Reduced management costs

They concluded that the principal beneficiaries also changed from local arable farmers under arable production to graziers, countryside users from towns and villages, and the global community, under restoration. However, it is not clear whether or not the value chain that the local arable farmers supported was included in the valuation.

Worrall et al. (2009) describe an opportunity to consider whether management practices in peatlands could be altered to enhance storage of C. The 'Sustainable Uplands' RELU project developed a model for calculating C fluxes from peat soils. The model was run for a decade from 1997-2006 and applied to an area of 550 km² of upland peat soils in the Peak District. The study estimated that management interventions could increase the total C sink overall and that a profit from C offsetting could be achieved within 30 years in areas where a C benefit was estimated. However, the area achieving a profit and the timescale was very dependent on the price of C used.

Using empirical evidence, Pretty et al. (2005) suggested that the opportunity cost per tonne of C stored for the majority of temperate agriculture exceeds US$50 t⁻¹. However, the final monetary value placed on a tonne of C will emerge either from a fully functioning C market or from government payment schemes. Estimates of the value of stored C have ranged from US$100 t⁻¹ to a low of less than US$5 t⁻¹. Pretty et al. (2005) suggested that the likely price was likely to be at the lower end of this range.

### 7.2.1 Implications for policy and further research

The evidence suggests that we are not yet in a position to assign an economic value to the ecosystem services that soils provide and that it may be more straightforward to assess soil quality in terms of the maintenance of soil functions and natural capital. One possible approach uses the concept of a “safe minimum standard” for soil quality and ES provision; and, where the standard is not met, the quantification of a ‘sustainability gap’, i.e. the degree to which soil quality and soil management practices need to be improved to enable delivery of multiple ES for current and future generations. However, for this to happen terms such as ‘soil quality’ and ‘sustainable’ need to be better defined for the key soil ES. It will also be important to separate the value of land from the value of soil, particularly within Natural Capital and environmental economic accounting assessments (Fraser and Robinson, 2015); and the indirect and non-marketable value of soils will also have to be taken into account (Table 20).

There will be an opportunity to fund projects that aim to better understand the complexities of the natural environment in valuation analyses and to consider the wider societal value of ecosystems services as part of the NERC-led “Valuing Nature” Programme; a six-year £7m
interdisciplinary research programme developed under the Living With Environmental Change partnership.

It may be more important and useful to monitor changes in soil properties and processes that provide fundamental ES than to try to assign an economic value to the specific ES that soils deliver. A growing challenge for soil science is to determine how it fits within the ecosystem services approach as relatively little thought has been given to soils (in relation to science, social science, and policy making).

It will be essential to continue to monitor more aspects of soil temporal change to help assess the impact of policy. The development of decision support tools (DST’s) could also be very helpful to inform policy. The integration of a soil ES approach and soil functions into DST’s could help policymakers assess the implications and tradeoffs associated with soil management policy decisions.

It will be important to consider:

- How to ensure that soil ES continue to be delivered into the future, i.e. that soils are managed sustainably. One approach could be to promote a diversity of agricultural systems by allocating land to the delivery of different goods and services according to its suitability. In other words, different parts of the landscape focusing on the delivery of particular goods and services as within-field patches and larger field or farm blocks.
- Better ways to set goals and objectives for soil quality that can be used in policy and management development, and for valuation.

7.3 Do we fully understand how soil management relates to soil function to accurately quantify this?

Volume and characteristics of the overall evidence base

The search and screening process, using key words related to the interactions between soil management and soil function, resulted in the allocation of only two papers to this research question. However, these two papers referenced 265 related documents and sources.

What does the evidence base indicate in relation to the question posed?

Stoate et al. (2009) provided an overview of the status of agricultural systems in Europe, including soil status. They and authors describe a number of relationships between soil management and soil function. For example, there is evidence that microbial biomass, especially that of soil fungi, is higher under reduced tillage than under plough-based cultivation systems, while earthworm densities can also be higher under reduced tillage (SOWAP, 2007). A number of authors have linked soil biota to specific soil functions such as nutrient and C cycling (e.g. Pulleman et al., 2012; de Vries et al., 2013). Soil cultivation tends to reduce populations and size of beetle larvae and can influence community structure by selecting against species with long-lasting larval stages (Ribera et al., 2001).

Physical disturbance through cultivation breaks up larger soil aggregates and exposes occluded C within aggregates to oxygen and biodegradation, allowing conditions for SOC
loss (Banwart et al., 2015). Soil disturbance and aeration factors that determine SOC accumulation and oxidation processes offer clues to soil management strategies that can maintain and increase soil C. SOC levels can therefore be maintained through controlling soil erosion, reducing the frequency/depth of cultivation, and increasing the water content of organic soils. Increasing C input can also be achieved by measures that increase biomass production and the partitioning of photosynthesis-derived C into roots and microbial biomass (Lal, 2010). The addition of imported organic matter such as compost can also impact on soil bio-physical properties, soil processes and function (Bhogal et al., 2009). However, it is also well established that the process of building up soil C and the associated soil resistance, resilience (to perturbation) and functionality is slow (Gregory et al., 2007; Gregory et al., 2009; Johnston et al., 2009).

Grasslands have numerous ecosystem functions and values: forage production, protecting soils from erosion and compaction and contributing to its fertility, water regulation, and potential for C sequestration (Hopkins and Holz, 2006). Agricultural grassland includes silage and hay fields, improved pastures for grazing, and semi-natural grasslands. As with arable systems, grasslands are sensitive to management with many positive functional attributes imparted by generally higher SOC levels. Nevertheless, grassland soils can still be degraded with impacts in managed grassland mainly associated with soil compaction due to high stocking rates and the increasing size of machinery (Newell Price et al., 2013).

Grassland management can also impact on plant diversity, which in turn can impact on functional diversity associated with the soil microbial community (de Vries et al., 2013). For example, in UK grassland an average of three forb species were found on livestock farms where N inputs were higher than 75 kg ha⁻¹, whereas higher forb diversity was found only in grasslands receiving less than 15 kg N ha⁻¹ (McCracken and Tallowin, 2004).

Iverson et al. (2014) introduced work conducted in eleven different countries on the contribution of landscape ecology to communicating the value and benefits from ES. The studies were designed to inform the economic, environmental and social values of the ecosystem services and should help to develop new management practices for sustaining ecosystems and the services they provide. The authors argue that potential new methods to evaluate integrated ecosystem services at the landscape level could offer new tools for developing soil management and conservation strategies.

7.3.1 Implications for policy and further research

**Arable**

- There is a need to better understand how soil management links to soil properties and soil function and the minimum standard of soil quality in terms of soil physical, chemical and biological properties and biological diversity (related to functional diversity and redundancy) needed to deliver the key soil ES.
- Soil compaction is a significant issue limiting soil function and ES delivery in many cultivated soils, and the development and testing of mitigation and avoidance methods needs greater emphasis in policy. The implications of soil compaction for soil processes and functions, or for flooding are poorly understood.
- An assessment of the relative benefits of reduced and no-till systems and the effects of increased herbicide use often associated with it needs further research.
• There is a need to better understand the recovery rates of soil C and the interactions between SOM, biodiversity, transformations of nutrients and structure and the physical stability of soil structure and soil aggregates; and to develop policies that encourage protecting, maintaining and enhancing soil C.

Grassland
• Recent changes to the CAP are likely to encourage more extensive grazing management in sheep and beef systems, but with a continuing trend towards intensification of dairy farming. These changes are likely to have both positive and negative impacts on soil quality and function depending on the location, nature and extent of the change and the farming system that is being altered.
• There is still considerable research required to understand how grassland management affects soil characteristics and nutrient cycling.
• More interdisciplinary research is needed to identify management regimes that support sustainable livestock production.

Energy crops
• The production of energy crops could potentially result in large scale and profound changes to agricultural landscapes. However, the likely extent of such changes and their impacts on ES such as nutrient cycling, C sequestration and flood control are poorly understood and require further study.

7.4 What work has been done to assess the relative cost and societal benefit of soil management interventions?

Volume and characteristics of the overall evidence base

The search and screening process resulted in 22 research papers that were related to assessing the relative cost and societal benefit of soil management interventions. Of the 22 papers identified, four covered the costing of management interventions or soil degradation; four considered environmental cost-benefit analysis or decision analysis in general; three discussed the potential for using payment for ecosystem services methods to provide an incentive or compensation for good practice; seven investigated the cost and societal benefit of land management practices in general; four did the same for soil management practices specifically; and one assessed the cost and marketable output (i.e. yield benefits) resulting from soil management interventions (Table 21).

Table 21. The number of papers that investigated the relative costs and/or benefits of soil management interventions by theme.

<table>
<thead>
<tr>
<th>Themes covered</th>
<th>Count</th>
</tr>
</thead>
<tbody>
<tr>
<td>Costing of management interventions or soil degradation/natural hazards</td>
<td>3</td>
</tr>
<tr>
<td>Environmental cost-benefit analysis or decision analysis</td>
<td>4</td>
</tr>
<tr>
<td>Payment for ecosystem services</td>
<td>3</td>
</tr>
<tr>
<td>Cost and societal benefit of land management practices</td>
<td>7</td>
</tr>
<tr>
<td>Cost and societal benefit of soil management practices</td>
<td>4</td>
</tr>
<tr>
<td>Cost and marketable output (only) resulting from soil management practices</td>
<td>1</td>
</tr>
</tbody>
</table>
Defra project SP1606 (The total costs of soil degradation in England and Wales) aimed to estimate the total economic cost of soil degradation in England and Wales, namely: erosion, compaction, decline in organic content, loss of soil biota, diffuse contamination and surface sealing. The results provide some indication of the societal benefit that could result from improved soil management to avoid soil degradation. The estimated total quantified costs of soil degradation in England and Wales were between £0.9 bn and £1.4 bn per year, with a central estimate of £1.2 bn. About 45% of total quantified annual soil degradation costs were associated with loss of organic matter, 39% with compaction and 13% with erosion.

The literature search identified few studies that explicitly assess the relative cost and societal benefit of soil management interventions. However, a number of papers provide suitable frameworks for this to be carried out. For example, Bouwer et al. (2014) costed measures for natural hazard mitigation in Europe; Wratten et al. (2012) and Breeze et al. (2014) used an expert appraisal approach to assess pollinator habitat enhancement and benefits to other ecosystem services; Beharry-Borg et al. (2013) evaluated farmers' likely participation in a payment for ecosystem services (PES) programme for water quality protection in the UK uplands; Wegner & Pascual (2011) considered “cost-benefit analysis in the context of ecosystem services for human well-being”; and Atkinson and Mourato (2008) stressed a need to understand when cost-benefit analysis is used in practice and why environmental decisions are often made in a manner apparently inconsistent with cost-benefit thinking.

Glenk and Colombo (2011) discussed issues related to the integration of soil C sequestration policies into agri-environment schemes and reported findings from a choice experiment to elicit preferences and estimate benefits of a soil C programme in Scotland. Benefit estimates suggested that including co-effects such as impacts on biodiversity and rural viability could significantly change the outcome of cost-benefit tests. Implications for the development of climate change policies were also discussed.

The cost and societal benefit of soil management measures was considered in Environment Agency project RM 830 on the “Identification of basic measures to address agriculture’s impact on water”. Implementation and ecosystem damage costs were compared based on £60 t⁻¹ carbon dioxide equivalent (CO₂e) (DECC, 2009), £2,100 t⁻¹ for ammonia-N (IGCB, 2008), £670 t⁻¹ for nitrate-N (Defra project WT0706), £35,000 t⁻¹ of phosphorus-P (Defra project WT0706), and £150 t⁻¹ for sediment (Defra projects WT0706 and WQ0106). Implementation of a suite of measures resulted in a mitigation benefit of between £1,400 and £3,800 at the farm scale and a cost-benefit ratio of between 1:1.75 and 2:1 (i.e. net societal benefit at the lower end of the interquartile range, but only one pound of ‘societal benefit’ gained from each two pounds spent on mitigation at the upper end). However, the saved damage costs associated with reduced microbial pathogen and biochemical oxygen demand (BOD) were not taken into account and would be in addition to these estimates.

The EU RECARE project (Preventing and remediating degradation of soils in Europe through Land Care) aims to quantify the impacts of soil degradation and conservation on soil functions and ecosystem services in a spatially explicit way, accounting for costs and
benefits, and possible trade-offs in 17 case studies. For example, case study 5 investigates the impact of soil compaction on soil functions and ES (Schjønning et al., 2015).

Bouwer et al. (2014) found that costing of management interventions in Europe has almost exclusively focused on estimating direct costs. They argue that “a cost assessment framework that addresses a range of costs, possibly informed by multiple stakeholders, would provide more accurate estimates and could provide better guidance to decision makers”.

In Saxony, Germany, Frank et al. (2014) assessed how soil erosion protection as a regulating service impacted six other ecosystem services, namely ecological integrity, provisioning services (provision of food and fibre, provision of biomass), regulating services (drought-risk regulation, flood regulation), and the cultural service of landscape aesthetics. They found synergies between three measures for reducing erosion (swales/grass waterways; hedgerows and no-till) and the provision of regulating and cultural services. In contrast, the impact on provisioning services was slightly negative. The integrated ES assessment approach, in combination with stakeholder involvement in the scenario development, helped communicate cross-sectoral effects of different management strategies in a comprehensive way and therefore supported regional planning.

Harrison et al. (2010) used literature review and expert elicitation to systematically document the importance of services and identify trends in their use and status over time for the main terrestrial and freshwater ecosystems in Europe. The condition of the majority of services provided by agricultural land showed either a degraded or mixed status across Europe with the exception of recent enhancements in freshwater provision and water/erosion regulation.

Societal benefit and the importance of different ES can depend on which stakeholder group is asked. In three European mountain regions, Lamarque et al. (2010) assessed differences between stakeholder groups in the identification of ecosystem services provided by grassland (which ecosystem services for whom), the relative rankings of these ecosystem services, and how stakeholders perceive the provision of these ecosystem services to be related to agricultural activities. The paper focused on the demand for ecosystem services rather than the factors that regulate ecosystem functions (i.e. the potential to deliver ecosystem services). They identified a common set of ecosystem services that were considered important by stakeholders across the three regions, including soil stability, water quantity and quality, forage quality, conservation of botanical diversity, aesthetics and recreation (for regional experts), and forage quantity and aesthetics (for local farmers) and conclude that practitioners, policy makers and researchers should be more explicit in their uses of the ecosystem services concept in order to be correctly understood and to foster improved communication among stakeholders.

MacLeod et al. (2010) developed a marginal abatement cost curve (MACC) to assess which crop and soil measures delivered the most economically efficient reductions in GHG’s in UK agriculture. An ‘efficient’ subset was identified with reference to a cost per tonne threshold of <=£100 t⁻¹ CO₂e. Similarly, Defra project SP1601 (Soil Functions, Quality and Degradation – Studies in Support of Implementation of Soil Policy) reviewed the overall costs and
benefits of soil erosion measures and concluded that the assessment of the cost-effectiveness of mitigation measures is hampered by a lack of data. There was little information available on the costs of the mitigation measures. Studies that assessed the effect of mitigation measures on soil loss mainly consider water quality, with other impacts rarely taken into account. Studies on the impacts of the mitigation measures on ecosystem services and their values were almost non-existent and therefore represent an important knowledge gap. Defra project SP1313 built on the results of SP1601 to assess the effectiveness, impact and cost of measures to protect agricultural soils specifically.

Meyer et al. (2013) provided an overview of the state-of-the-art approaches to costing natural hazards such as drought and floods and discussed key knowledge gaps and uncertainties. They showed that the application of cost assessments in practice is often incomplete and biased, as direct costs receive a relatively large amount of attention, while intangible and indirect effects are rarely considered. Recommendations addressed how risk dynamics due to climate and socio-economic change can be better considered, and in what ways cost assessment can function as part of decision support.

Stewart et al. (1998) assessed the economic consequences of using novel wheel-traffic systems, zero (gantry or spanner) and reduced ground pressure, to minimise compaction, using crop input and output data from long-term studies of arable rotation cropping and grass grown for silage. For barley and oilseed rape crops, enterprise profitability was maintained with a zero traffic system and for grassland, gross margins were higher from both novel systems than from the conventional traffic system. By contrast, for potatoes, there was no benefit from either of the novel traffic systems. However, they did not assess the wider societal benefits of the methods.

Similarly, Chamen et al. (2015) reviewed and assessed the costs and benefits of soil compaction mitigation and avoidance methods on UK soil types. For the mitigation options assessed, only targeted sub-soiling resulted in a positive change to gross margin, between £0 ha⁻¹ for sandy soil and £22 ha⁻¹ for clay soil. All soil compaction avoidance technologies increased gross margins significantly, ranging from £26 ha⁻¹ for tracked tractors on sandy soil to £118 ha⁻¹ for controlled traffic farming (CTF) on clay soil. The impact of soil compaction on ecosystem services provided by soil were also considered. Avoidance technologies improve soil structure, increase rainfall infiltration and emissions of N, providing a ‘win-win’ situation for farmers and ecosystem services.

In Wales, monitoring of the Glastir agri-environment scheme has included an assessment of the ecosystem service benefits derived from the scheme (e.g. Wynne-Jones, 2013), including analysis of the societal benefits of soil C storage and rural landscapes.

Peatlands provide a number of ecosystem benefits, including storing C (e.g. Scotland’s peatlands contain c. 1,600 tonnes of C); supporting biodiversity; purifying and retaining water; preserving archaeological artefacts; and providing social and economic benefits through tourism and recreational uses. Dobbie et al. (2011) emphasised the need to better understand the extent and condition of peatlands to help ensure their adequate protection and restoration (JNCC, 2011). Defra project SP0572 also assessed the distribution and cost-benefit flows of different ecosystem services in upland and lowland peatlands using upland
and lowland peatland demonstration case study sites including the Moors for the Future Partnership.

7.4.1 Implications for policy and further research

The evidence indicates the importance of compaction avoidance, SOC enhancement and vegetation cover strategies in agriculture. However, more evidence is needed for different soil types, farm types and agro-climatic regions to better inform farmers and advisers of best soil management practices in different circumstances.

More work is needed on the costs of soil management interventions that go beyond the simple assessment of direct costs and include intangible and indirect effects. There is also a need for more information on societal benefits including the development of ecosystem damage costs for a wider range of pollutants, goods and services.

New methods are required for assessing the full range of societal benefits from soil management interventions from replacement cost to willingness to pay, stated preferences and the full market and non-market value of ecosystem goods and services. In order to promote, quantify and evaluate PES initiatives, it will be important to develop soil quality indicator change values as proxies for soil-related ecosystem service and natural capital benefits.

It will be important to consider how ES concepts and societal benefit from soil management policies can be better communicated to the public so that policy is correctly understood and communication between different stakeholder groups is improved.
8. Evidence gaps and priorities for new research

8.1 Soil quality indicators and ecosystem services

More research is needed to assess the detectable degree of change in soil properties and the change in the proportion of soils within particular SQI value ranges that may indicate soil degradation and a trend towards loss of function. Interpretation of indicator values in terms of ES delivery is a first step in using indicators in soil monitoring programmes to support decision making by land managers and policy makers.

There is a need to provide robust scientific evidence to explore relationships between soil properties and soil function through randomised and replicated field experiments. Long term experiments are essential for determining the relationship between SQI values, soil management and ES delivery. Detailed investigation such as that carried out at the Sourhope Research Station (as part of the NERC project on “Biological Diversity and Function in Soils”), Defra’s Soil QC (SP0530) and Long-term Biosolids sites (SP0143) should be extended to other sites to improve our understanding of C transformations; nutrient cycles; soil structure maintenance; and the regulation of pests and disease (i.e. soil health and sustainability) and how these relate to ES delivery.

It will be important to develop indicator reference values for different combinations of land use, soil type and climate to determine a meaningful degree of difference in indicator values that represents actual change in terms of ES delivery (Pulleman et al., 2012). Such references do not yet exist for many indicators at a national scale.

In policy terms, a key issue is where the balance is struck between production and the provision of other ES goods and services. There is strong evidence that to truly optimise the soil physical conditions and soil biota communities for particular ES will require context-dependent approaches. Each individual soil cannot provide all functions to an optimal level simultaneously, and so preference needs to be given to certain ecosystem goods and services depending on the context and decision-support tools or frameworks could be developed to guide this process.

Given that the maintenance of SOC concentrations is a vital component of sustainable agricultural systems and that one way of achieving this is to sequester C through cover cropping (e.g. Defra project SP1106A - Quantification of the potential changes in soil carbon in England from soil protection Measures within the Soil Protection Review 2010) it could be worth developing indicators of soil management that help prevent organic matter loss and associated soil degradation. It will be important to investigate the potential of observational technologies (e.g. earth observation, mapping, sensors) and analysis technologies (e.g. modelling, big data and analytics), and by combining observations with models to improve predictive potential for sustainable soil management in the UK.

The potential use of public participation in the measurement of certain suitable SQI’s merits further investigation since such initiatives can provide large and useful datasets which not only contribute towards the soil monitoring evidence base, but also engage different stakeholder groups into considering the importance of soils and their management.
There are a number of new emerging indicators (such as the measurement of soil aggregate stability using laser granulometry; the quantification of soil structure using spectral reflectance, image processing technology and X-ray micro computed tomography; and the use of enzyme activity as an early indicator of soil structural degradation) for which typical reference values and ranges need to be established for different combinations of soil type, land use and management. These values also need to be related to soil processes, functions and ES delivery.

For organo-mineral soils in particular there is a lack of field measurements of the potential C storage benefits that are possible under different land use cover types; and considerable uncertainty in the timescale over which such benefits can be attained.

To improve our understanding of soil health and sustainability and how to measure them, the following research areas should be explored:

- More field research is needed to relate soil management to SQI values and ES outcomes (e.g. crop quality and yield in the case of food production) and to develop effective conceptual models of the relationship between SQIs, soil functions and the delivery of ecosystem goods and services across a range of soil types, land uses and management scenarios and at different points in the soil life cycle (formation, productive use and degradation).
- Investigation into the nature of C and nutrient cycling in a range of soil types in productive agricultural systems and the long term effects of agricultural inputs on these soil processes.
- Better understanding of soil C dynamics will be key to achieving sustained delivery of soil ecosystem services.
- An important step in the process will be to validate existing conceptual models with new and existing experimental evidence.
- Better understanding of soil biodiversity and its importance for soil processes and functions for different land use, soil management and soil type combinations
  - Functional diversity and functional redundancy within different agricultural systems
  - Links between soil biota diversity, functional diversity and soil morphology at various scales
- The use of enzyme activity as an early indicator of soil structural degradation.
- The use of visible (VIS) and near-infrared (NIR) spectroscopy for prediction of soil properties and soil structural quality.
- The development of standard operating procedures including calibration and guidance on adequate replication, and protocols need to be developed for new indicators.
- The nature of SOC, particularly the 'light' or 'fresh' fraction and its influence on soil properties under different land uses.
- More quantitative evidence to develop critical thresholds and workable ranges for SOC for different soil types, land uses and ecosystem services.
- The implications of manufactured fertiliser use for C sequestration and storage and the balance between climate change mitigation considerations and food production.
- The key factors and processes influencing C fluxes in peatlands.
• The functional ecology of AMF and other soil biota in different agricultural systems and possible synergies and antagonisms with a need for sustainable intensification.
• The importance of fungal diversity in C and nutrient cycling.
• How soil biodiversity is related to the provision of multiple ecosystem services.

8.2 Soil degradation

Soil erosion

Further research is needed to provide additional information on both baseline soil erosion rates (including further field-based monitoring) and the efficacy of mitigation strategies across a wider range of locations, given the large spatial-temporal variation reported by many studies. Resources should focus more on monitoring soil erosion by field measurements over modelling.

There is a need to better understand how the critical soil erosion threshold of 1 t ha\(^{-1}\) yr\(^{-1}\) (Verheijen et al., 2012) relates to ecosystem functioning; and for soil erosion mitigation strategies that can be integrated into sustainable agricultural systems in high risk areas (Broadman and Favis-Morlock, 2014).

Numerical models of soil formation should be developed to help implement soil erosion mitigation strategies at appropriate spatial scales and in appropriate locations.

Soil organic matter decline

We need to investigate whether or not there are SOC concentration critical thresholds for the delivery of specific ES in different circumstances or whether ecosystem function is determined by other factors not closely correlated with SOC concentration.

Given that organic soils are sensitive to temperature, with organic C declining at temperatures above 7°C, future monitoring of SOC stocks should focus on soils with soil C concentrations above 250 and 435 g kg\(^{-1}\) (Barraclough et al., 2015).

N deposition - acidification

Further research is required to assess the impact of N deposition on species-rich grassland under different management strategies.

Further work is required “to establish the impacts of changing soil acidity on the wider C cycle and to ensure that observed changes in terrestrial C cycling, particularly those based on measurements in industrialised regions, are not erroneously attributed to other drivers” (Evans et al., 2012).

Soil compaction

Further work is needed to investigate the extent to which soil compaction is limiting ecosystem function and ES delivery; and on innovative and practical mitigation strategies. The investigations should include an assessment of structural changes in the agricultural industry that may be needed to optimise soil structure in arable and grassland systems.
Monitoring of soil compaction (soil structure and bulk density) could help determine whether any changes in the nature and timing of rainfall, and its relationship with soil management practices, is affecting the severity of soil structural degradation.

8.3 Sustainable soils / Aspirational soil quality targets
There is a clear need to define specific criteria for soil sustainability depending on the ecosystem service in question and to define concepts such as ‘soil health’ and ‘sustainable’ in order to set goals and objectives that can be used in policy and management development.

There is a continued need to develop rapid methods of soil structural assessment from practical visual evaluation methods that can be used in the field by practitioners to the development of new technology to provide research tools that are more effective at quantifying soil structure on a continuous scale. The preservation of soil structure is “key to sustaining soil function” (Mueller et al., 2010).

There is also a continued need to investigate the implications of soil management practices, particularly the input of organic materials and manufactured fertilisers, for soil metal concentrations.

It will be important to develop models such as MOSES and the CZO models to include additional processes and functions to improve the simulation of multiple soil ecosystem service provision and the effects of external drivers such as climate change. This will be an important area for development to aid decision making and help define what we mean by a sustainable soil in order to set specific goals and objectives for policy.

There is a continued need for stakeholder engagement and interdisciplinary research to help address land-based challenges. Transparency of scientific information and the early identification of synergies, conflicts and trade-offs between different objectives and stakeholder groups is essential to reconcile concerns and agendas. This emphasises the need to translate scientific research findings into clear advisory tools so that all stakeholder groups can engage in discussion about soil management practices and sustainability.

It will be important to extend detailed assessments of the biological diversity and functional diversity to a wider range of agricultural soils to determine whether or not current management practices are sustainable in terms of their ability to deliver the key ecosystem services.

Further work is required on:

- The merits and risks of earthworm introduction into fields and other forms of soil biota/food web manipulation.
- Developing a methodology to quantify the risk of harm to soil by soil organic matter decline.
- Establishing acceptable rates of change in key soil properties such as bulk density, SOC and functional groups of soil biota for different combinations of soil type, land use, climate and ES.
8.4 Soil management practices
Further work is needed to:
- Update existing soil C models to take account of a better understanding of SOM dynamics.
- Produce crop-specific and environment-specific C input values to improve the reliability of SOC stock predictions.
- Collate a substantial body of research in order to assess how different broad soil types (e.g. mineral, organo-mineral and peaty) respond to soil management practices within arable and grassland systems.
- Provide more quantitative information on how C is apportioned between different microbial functional groups to inform whether or not the plant-soil system can be manipulated to favour organisms that promote soil C storage in agricultural soils.
- Assess the C sequestration potential of manipulating plant species diversity and associated soil biodiversity in grassland soils and implications for overall food production levels.
- Carry out pilot-level field work, to examine the feasibility of, and reduce the uncertainties in, biological mitigation strategies.
- Engage with the public concerning mitigation strategies as certain biological approaches may have negative effects on some ecosystem services and land use (Woodward et al. 2009).
- Develop techniques for up-scaling local interventions to quantify landscape-scale benefits.
- Investigate the social, economic and environmental viability of new climate change mitigation and adaptation techniques such as paludiculture.
- Investigate the potential impacts of changes in temperature and seasonal rainfall patterns (under a range of potential climate change scenarios) on whole ecosystem functioning in grassland and arable soils to provide a better understanding of longer-term impacts.
- The development of more comprehensive process-based models combining mathematical relationships and expert opinion to provide a general assessment of the effects of farm mitigation and adaptation on environmental losses under a changing climate.
- More integrated approaches to climate impact assessments including consideration of socio-economic aspects of climate change adaptation.
- In Scotland’s National Peatland Plan (SNH consultation paper, 2014) the key priorities for research include: the current state and extent of restorable peatland in Scotland, the impacts of restoration on net greenhouse gas emissions, the impacts of N and S deposition on peat forming vegetation e.g. Sphagnum mosses, approaches to maximise the ecosystem benefits across C, biodiversity and water by restoring degraded peatlands and maintenance of healthy peatlands.

8.5 Economic value of ecosystem services
It will be essential to continue to monitor more aspects of soil temporal change to help assess the impact of policy. The development of decision support tools (DST’s) could also be very helpful to inform policy. In addition, more evidence is needed of the effectiveness of sustainable soil management practices, such as compaction avoidance, SOC enhancement
and vegetation cover strategies on different soil types, farm types and agro-climatic regions to better inform farmers and advisers of best soil management practices in different circumstances.

**Arable**
- We need a better understanding of how soil management links to soil properties and soil function and the minimum standard of soil quality in terms of soil physical, chemical and biological properties and biological diversity (related to functional diversity and redundancy) needed to deliver the key soil ES.
- Further development and testing of soil compaction avoidance and mitigation methods is needed, since the implications of soil compaction for soil processes and functions, or for flooding are poorly understood.
- An assessment of the relative benefits of reduced and no-till systems and the effects of increased herbicide use often associated with it needs further research.
- We need to better understand the recovery rates of soil C and the interactions between SOM, biodiversity, transformations of nutrients and structure and the physical stability of soil structure and soil aggregates; and to develop policies that encourage protecting, maintaining and enhancing soil C.

**Grassland**
- There is still considerable research required to understand how grassland management affects soil characteristics and nutrient cycling.
- More interdisciplinary research is needed to identify management regimes that support sustainable livestock production.

**Energy crops**
- The impacts of increased energy crop production on ES such as nutrient cycling, C sequestration and flood control are poorly understood and require further study.

More work is needed on the costs of soil management interventions that go beyond the simple assessment of direct costs and include intangible and indirect effects. There is also a need for more information on societal benefits including the development of ecosystem damage costs for a wider range of pollutants, goods and services.

New methods are required for assessing the full range of societal benefits from soil management interventions from replacement cost to willingness to pay, stated preferences and the full market and non-market value of ecosystem goods and services.

It will be important to consider how ES concepts and societal benefit from soil management policies can be better communicated to the public so that policy is correctly understood and communication between different stakeholder groups is improved.
9. Summary and Conclusions
Evidence was gathered using ‘Web of Science’ searches using combinations of agreed keywords related to five themes:

- Soil quality indicators and ecosystem services
- Soil degradation
- Sustainable soils
- Sustainable soil management practices
- Economic value of ecosystem services

Searches were restricted to papers from UK-affiliated research organisations published since 1995 resulting in 1,563 papers. In a first screening, titles were then checked for relevance to specific research questions and then abstracts checked in a second screening resulting in the identification of 559 peer reviewed research papers in total. Key papers were also identified for more detailed assessment. Quick searches were carried out to identify relevant projects that addressed the research questions and themes using Defra, BBSRC, NERC, ESRC, Forestry Research and Scottish Government search facilities.

The review and themes respond to the main aim of the Soil Security Programme to improve understanding of how soil systems respond to changes in management and climate and Defra’s target that all soils should be managed sustainably and degradation threats tackled successfully by 2030. The research questions for the evidence review included:

- How can soil quality indicator (SQI) values be interpreted with respect to ecosystem service delivery?
- At what point do soil degradation processes significantly affect soil quality and function?
- What properties should a sustainable soil have?
- What effect may climate change have on aspirational soil quality targets?
- How should soil be managed to achieve sustainability?
- Can soil biodiversity be manipulated to improve ecosystem service delivery?
- What is the economic value of the key ecosystem services provided by sustainably managed soils?

In July 2015, the findings were discussed at three workshops in Edinburgh, Aberystwyth and Reading to provide an opportunity for researchers, advisers and other relevant stakeholders to provide feedback on draft review outputs, validate the research gaps identified and help determine the direction of the Soil Security Programme.

The evidence review indicates that the relationship between SQI values and ecosystem service delivery is still poorly understood. Thresholds and workable ranges for physical and chemical indicators have been developed, but not necessarily related to the delivery of specific soil functions in different circumstances. Soil biological indicators require further investigation to better understand their capacity to reflect the degree of soil degradation or the ability of soils to perform specific functions. The development of reference values for particular soil type, land use and climate combinations could help provide a useful framework against which soil quality could be assessed.
For soil degradation, more work is needed on assessing soil erosion in high risk areas, including field scale monitoring and evaluation of the efficacy of different mitigation options. The impact of soil compaction on ES is also poorly understood and further work is needed to assess compaction avoidance and mitigation techniques for integration into farming systems. Soil organic matter decline has significant implications for soil function and climate change mitigation and we need to understand C cycling dynamics better particularly in organo-mineral and peat soils. The impacts of changing soil acidity on ES in the uplands needs further research.

For most ecosystem services, we do not have a clear idea of what properties a sustainable soil should have in terms of (for example) SOC content, soil structure, nutrient content and metal concentrations, although past research outputs provide some guidance. Sustainability also needs to be considered in terms of what ES should be prioritised in different parts of the landscape and what sustainability means for each ES in each location (i.e. combination of soil type, slope, land use and climate).

There is a wealth of information on how soil management practices impact on soil properties and crop production, but for other ES there is less information. We also have a poor understanding of whether or not soil biodiversity can be manipulated to improve soil function. It is clear that soils have the capacity to store more C, but there are constraints to the ability of this approach to provide genuine C sequestration and climate change mitigation. However, strategies that promote the maintenance or enhancement of SOC will generally improve soil properties for multiple benefits.

The development of tools that incorporate the economic value of ecosystem services provided by soils is in its early stages. Only a few studies have attempted to value soil ES rather than simply surveying prices of soils as a marketable commodity. However, there is potential for the valuation of soil to be used to develop decision support tools to support resource management and soil protection policy. The development of such valuation tools could help support policy mechanisms such as Payments for Ecosystem Services (PES).

The evidence review and workshop discussion resulted in the following recommendations for future research:

9.2 Soil quality indicators and ecosystem services
- Soil C and nutrient dynamics in a range of soil types in productive agricultural systems and the long term effects of agricultural management on these soil processes.
- Field research and continued long-term field experiments on the relationship between soil management, SQI values and ES outcomes in a range of soil types, land uses and management scenarios.
- The development of standard operating procedures including calibration and guidance on adequate replication, and protocols need to be developed for new SQI’s.
- The nature of SOC, particularly the ‘light’ or ‘fresh’ fraction and its influence on soil properties and soil function under different land uses.
- The key factors and processes influencing C fluxes in peatlands.
• Better understanding of soil biodiversity and its importance for soil processes and functions for different land use, soil management and soil type combinations.
• The functional ecology of AMF and other soil biota in different agricultural systems and possible synergies and antagonisms with a need for sustainable intensification.
• The importance of fungal diversity in C and nutrient cycling.
• The implications of manufactured fertiliser use for C sequestration and storage and the balance between climate change mitigation considerations and food production.

9.3 Soil degradation

Soil erosion
• Baseline soil erosion rates and the efficacy of mitigation strategies across a wider range of locations.
• Focus on soil erosion monitoring by field measurements to inform models.
• The integration of practical soil erosion mitigation strategies into sustainable agricultural systems in high risk areas.

Soil organic matter decline
• Quantitative evidence to develop critical thresholds and workable ranges for SOC for different soil types, land uses and ecosystem services or is ecosystem function determined by other factors not closely correlated with SOC concentration?
• Further monitoring of SOC stocks with focus on soils with soil C concentrations above 250 and 435 g kg\(^{-1}\).

N deposition - acidification
• The impact of N deposition on species-rich grassland under different management strategies.
• Establish the impacts of changing soil acidity on the wider C cycle.

Soil compaction
• The extent to which soil compaction is limiting ecosystem function and ES delivery.
• Development of practical mitigation strategies that may be needed to optimise soil structure in arable and grassland systems.

9.4 Sustainable soils
• The merits and risks of earthworm introduction into fields and other forms of soil biota/food web manipulation.
• Developing a methodology to quantify the risk of harm to soil by soil organic matter decline.
• Establishing acceptable rates of change in key soil properties such as bulk density, SOC and functional groups of soil biota for different combinations of soil type, land use, climate and ES.

9.5 Sustainable soil management practices
• Investigate how different broad soil types (e.g. mineral, organo-mineral and peaty) respond to soil management practices within arable and grassland systems.
• Further development and testing of soil compaction avoidance and mitigation methods.
• Quantitative information on how C is apportioned between different microbial functional
groups to inform whether or not the plant-soil system can be manipulated to favour
organisms that promote soil C storage in agricultural soils.
• Update existing soil C models to take account of a better understanding of SOM
dynamics.
• Pilot-level field work, to examine the feasibility of, and reduce the uncertainties in,
biological mitigation strategies.
• Investigate the social, economic and environmental viability of new climate change
mitigation and adaptation techniques such as paludiculture.
• More integrated approaches to climate change adaptation including consideration of
socio-economic aspects.

9.6 Economic value of ecosystem services
• The development of decision support tools (DST’s) that put a value on specific aspects of
soil quality and the delivery of ES could help inform policy.
• Better understanding of how soil management links to soil properties and soil function
and the minimum standard of soil quality in terms of soil physical, chemical and biological
properties and biological diversity (related to functional diversity and redundancy)
needed to deliver the key soil ES.
• An assessment of the relative benefits of reduced and no-till systems and the effects of
associated increases in herbicide use.
• Investigations into the recovery rates of soil C and the interactions between SOM, soil
biodiversity, transformations of nutrients and structure and the physical stability of soil
structure and soil aggregates.
• Development of policies that encourage protecting, maintaining and enhancing soil C.
• To better understand how grassland management affects soil characteristics and nutrient
cycling.
• Identification of management regimes that support sustainable livestock production.
• The costs of soil management interventions that go beyond the simple assessment of
direct costs and include intangible and indirect effects.
• More information on the societal benefits of soil management interventions including the
development of ecosystem damage costs for a wider range of pollutants, goods and
services.
• How ES concepts and societal benefit from soil management policies can be better
communicated to the public so that policy is correctly understood and communicated
between different stakeholder groups.